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EBASCO SERVICES INCORPORATED

DRAFT FINAL

BASELINE ECOLOGICAL RISK ASSESSMENT NEW BEDFORD HARBOR SITE FEASIBILITY STUDY

APRIL 1990

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ECOLOGICAL RISK ASSESSMENT

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EXECUTIVE SUMMARY

New Bedford Harbor is a tidal estuary on the western shore of Buzzards Bay, Massachusetts. Studies of the harbor conducted in the mid-1970s indicated widespread polychlorinated biphenyl (PCB) and heavy metals contamination. Large areas of the harbor were subsequently closed to fishing to reduce the potential for human exposure to PCBs. The New Bedford Harbor site was added to the U.S. Environmental Protection Agency (EPA) Interim National Priorities List in July 1982; shortly thereafter, EPA initiated a more comprehensive assessment of the extent of the PCB contamination problem. These and other studies have confirmed extensive PCB contamination of water, sediments, and biota in the harbor, with sediment concentrations reported in excess of 100,000 parts per million (ppm) in the area of maximum contamination. Concentrations in biota in many areas exceed the U.S. Food and Drug Administration tolerance level of 2 ppm.

Under authority of the Comprehensive Environmental Response, Compensation, and Liability Act (or Superfund), EPA is responsible for conducting a Remedial Investigation and Feasibility Study (RI/FS) to support the need for and extent of remediation in New Bedford Harbor. This baseline ecological risk assessment, as part of the RI/FS process, presents and quantifies risks to aquatic organisms due to exposure to PCBs and heavy metals in New Bedford Harbor. Based on current conditions in the harbor, it will serve as a benchmark against which the effectiveness of various remedial options may be evaluated.

The ecological risk assessment is based on data collected by several investigations, but draws most heavily on information generated by Battelle (Battelle Pacific Northwest Laboratories, Richland, Washington; and Battelle Ocean Sciences, Duxbury, Massachusetts) in conjunction with the development of a numerical hydrodynamic/sediment-transport model of the harbor. Risk to aquatic biota was evaluated using a joint probability analysis in which two probability distributions, one representing contaminant levels in various zones of the harbor and the second representing the sensitivity of biota to contaminants, were combined to present a comprehensive probabilistic evaluation of risk. The joint probability analysis was supplemented by comparison of PCB levels in the harbor to EPA water quality criteria, evaluation of site-specific toxicity tests, and examination of data on the structure of faunal communities in the harbor.

Results of these various approaches to evaluating risk, both together and independently, support the conclusion that aquatic organisms are at significant risk due to exposure to

PCBs in New Bedford Harbor. Some risk due to exposure to metals was also identified; however, it was negligible compared to the risk due to PCBs.

Concentrations of dissolved PCBs in the area of maximum contamination (i.e., the Hot Spot) and in all areas of the Inner Harbor (i.e., inside the Hurricane Barrier) were sufficiently elevated to result in a significant likelihood of chronic effects to indigenous biota. PCB concentrations in sediment and sediment pore water in many areas of the harbor were found to be highly toxic to at least some members of all major taxonomic groups of organisms. In the Upper Estuary, the probability of these sediments being toxic to marine fish, the most sensitive taxonomic group investigated, approached certainty. These conclusions were found to be consistent with the reported results of laboratory experiments conducted using New Bedford Harbor sediments and with available data on faunal community structure. EPA ambient water quality criteria and interim sediment quality criteria were exceeded in many areas of the Inner Harbor.

Potential community or ecosystem level impacts due to PCBs in New Bedford Harbor cannot be evaluated fully by assessing impacts to individual species or taxonomic groups. However, the state of development of ecological risk assessment methodology does not allow quantification of impacts or risk at these higher levels. Nonetheless, the results of numerous site-specific and laboratory studies, including this risk assessment, indicate that New Bedford Harbor is an ecosystem under stress and there is a high probability that PCBs are a significant contributing factor to the integrity of the harbor as an integrated functioning ecosystem.

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1.0 INTRODUCTION

1.1 NEW BEDFORD HARBOR ECOSYSTEM

New Bedford Harbor is a tidal estuary on the western shore of Buzzards Bay, Massachusetts, situated between the City of New Bedford on the west and the towns of Fairhaven and Acushnet on the east. The area contains approximately six square miles of open water, tidal creeks, salt marshes, and wetlands. The major freshwater inflow to this area is the Acushnet River, a small stream with mean annual flow of approximately 1 cubic meter per second. As a result, the system does not fit the traditional definition of an estuary; salinities throughout the harbor are high and the strong horizontal and vertical salinity gradients that control patterns of faunal distribution in estuaries are absent. Nonetheless, the system does provide habitats for a wide variety of aquatic organisms that use this area for spawning, foraging, and overwintering.

The topographical characteristics of New Bedford Harbor have been adequately described in several other reports generated as a result of studies undertaken to provide information for the Remedial Investigation/Feasibility Study (RI/FS) process and will not be repeated herein. However, several features of the area have importance for understanding the ecological risk assessment. The estuary and harbor may be conveniently divided into subareas by bridges and other manmade structures that also represent logical divisions between zones of ecological similarity. Therefore, the Coggeshall Street Bridge represents not only a convenient boundary for the area defined in these studies as the Upper Estuary, but also separates an area of shallow water with predominantly organic silts and clays with silty sands poorly sorted muddy to the north from deeper water with silty sands to the south (Figure 1-1). At the State Route 6 Bridge (Popes Island), depths generally increase, with water depths in most of the area south of the bridge maintained by dredging. This area of New Bedford Harbor is also the most heavily impacted by industrialization, with considerable shoreline development and ship traffic related to the fishing industry.

The Lower Harbor ends at the Hurricane Barrier, which separates the comparatively low-energy silty sediment of the harbor from the high-energy sands typical of littoral areas in Buzzards Bay. The Hurricane Barrier represents a significant feature of importance for the current regime in the harbor, and the jet effect created by the narrow opening dominates patterns of mixing.

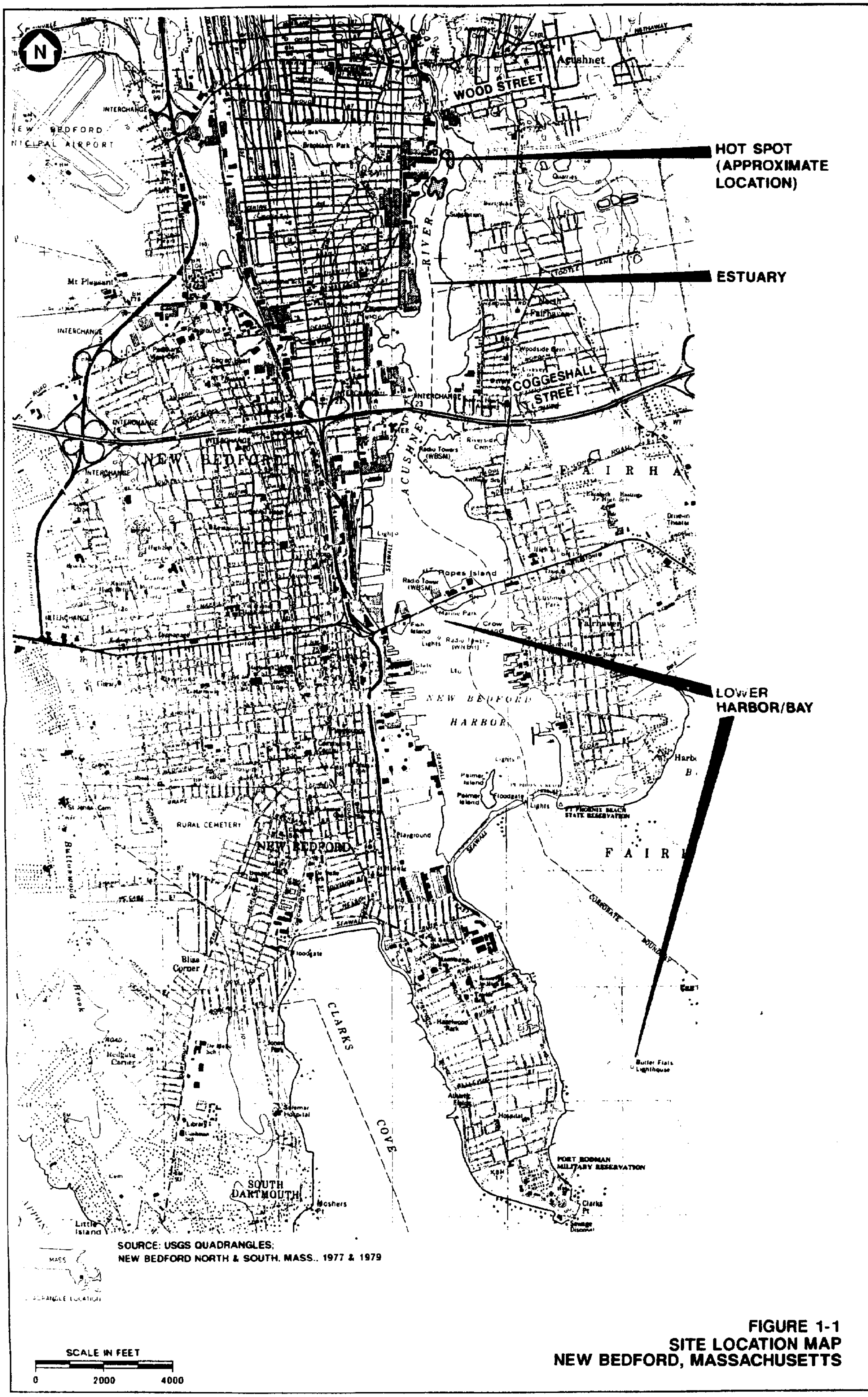


FIGURE 1-1
SITE LOCATION MAP
NEW BEDFORD, MASSACHUSETTS

1.2 SITE HISTORY

Between 1974 and 1982, a number of environmental studies were conducted to assess the magnitude and distribution of polychlorinated biphenyl (PCB) and, to a lesser extent, heavy metals contamination in New Bedford Harbor. Results of these studies revealed that sediment north of the Hurricane Barrier contain elevated levels of PCBs and heavy metals. Additional investigations revealed that PCBs had been discharged into the surface waters of New Bedford Harbor, causing significantly elevated PCB concentrations in sediment, water, fish, and shellfish.

To reduce the potential for human exposure to PCBs, the Massachusetts Department of Public Health closed much of the New Bedford Harbor area to fishing. Three closure areas were established on September 25, 1979 (Figure 1-2). Area 1 (New Bedford Harbor) is closed to the taking of all finfish, shellfish, and lobster. Area 2 (Hurricane Barrier to a line extending from Ricketson Point to Wilbur Point) is closed to the taking of lobster and bottom-feeding fish (eel, scup, flounder, and tautog). Area 3 (from Area 2 out to a line from Mishaum Point, Negro Ledge, and Rock Point) is closed to the taking of lobster.

In July 1982, the U.S. Environmental Protection Agency (EPA) placed New Bedford Harbor on the Interim National Priorities List (NPL). The final NPL was promulgated in September 1984. The site, as listed, includes the Upper Estuary of Acushnet River, New Bedford Harbor, and portions of Buzzards Bay. Following the NPL listing, EPA Region I initiated a comprehensive assessment of the PCB problem in the New Bedford Harbor area, including an areawide ambient air monitoring program, sediment sampling in the Acushnet River and New Bedford Harbor, and biota sampling in the estuary and harbor.

As a result of these studies, the extent of PCB contamination is better understood. The entire harbor north of the Hurricane Barrier, an area of 985 acres, is underlain by sediment containing elevated levels of PCBs and heavy metals. PCB concentrations in this area range from a few parts per million (ppm) to more than 100,000 ppm. Portions of western Buzzards Bay sediment are also contaminated, with PCB concentrations occasionally exceeding 50 ppm. The water column in New Bedford Harbor has been measured to contain PCBs in excess of the EPA 30-parts-per-trillion ambient water quality criterion

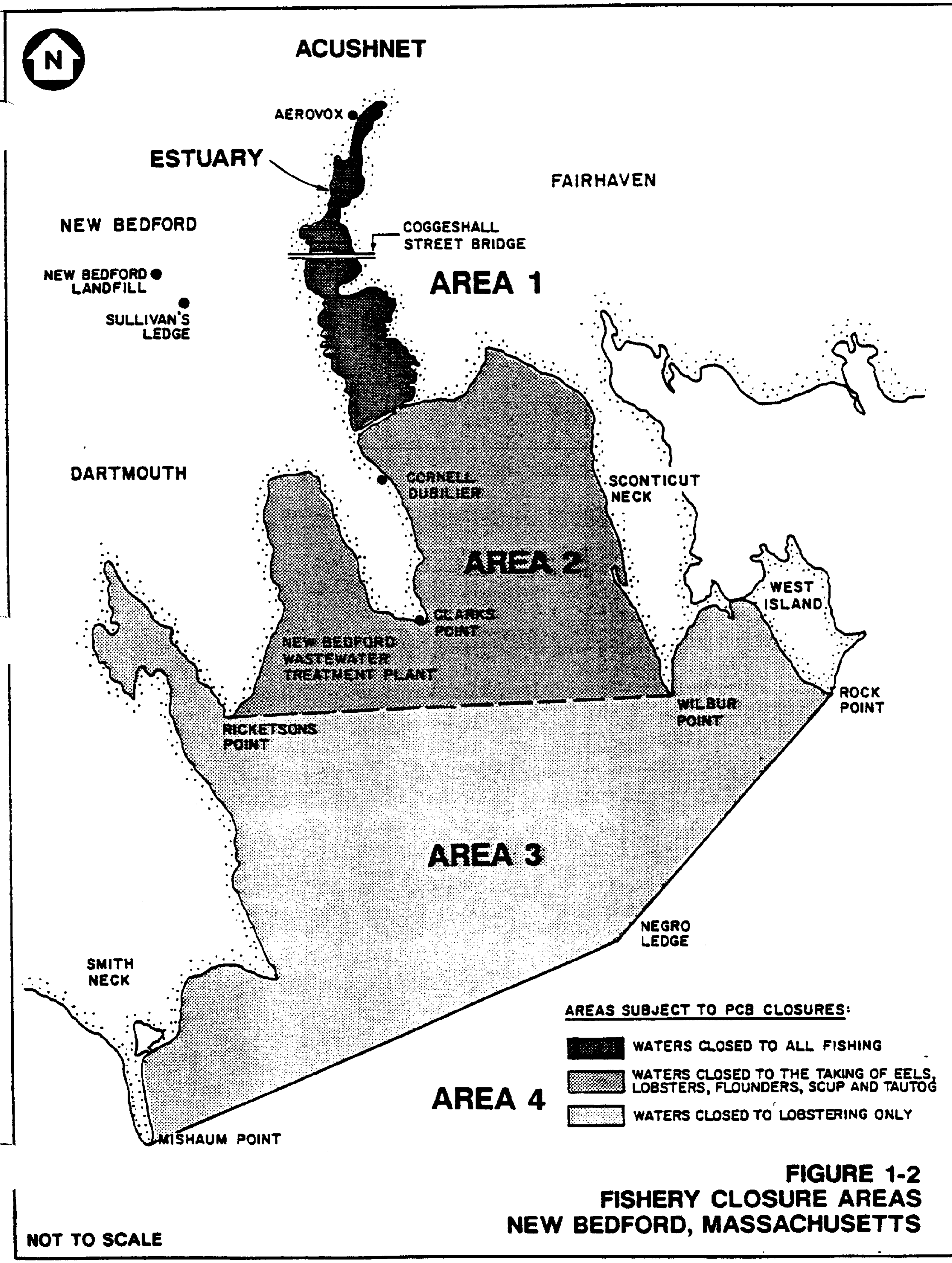


FIGURE 1-2
FISHERY CLOSURE AREAS
NEW BEDFORD, MASSACHUSETTS

(AWQC). Concentrations of PCBs in edible portions of locally caught fish have been measured in excess of the U.S. Food and Drug Administration (FDA) 2-ppm tolerance level for PCBs.

In 1984, EPA conducted an initial FS of the highly contaminated mudflats and sediment in the Upper Estuary of Acushnet River (NUS, 1984a and 1984b). Five clean-up options were presented in that report. EPA received extensive comments on these options from other federal, state, and local officials, potentially responsible parties, and the public. Many of the comments expressed concern regarding the proposed dredging techniques and potential impacts of dredging on the harbor, and potential leachate from the proposed unlined disposal sites.

In responding to these comments, EPA elected to conduct additional studies before choosing a clean-up alternative for the Upper Estuary. Concurrent with these studies, EPA conducted additional surveys to better define the extent of PCB contamination throughout the overall harbor and bay. Through these efforts, clean-up options for the site are being developed.

1.3 OBJECTIVES AND LIMITATIONS OF THIS REPORT

EPA Region I is responsible for the cleanup of the New Bedford Harbor site under authority of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) as amended by the Superfund Amendments and Reauthorization Act of 1986. Pursuant to this charter, EPA has direct responsibility for conducting the appropriate studies for this site to support the need for and extent of remediation. In accordance with the National Contingency Plan, these studies form the basis of the RI/FS for the site.

This ecological risk assessment presents and quantifies risks to aquatic organisms due to exposure to PCBs and selected heavy metals (i.e., copper, cadium, and lead) in the New Bedford Harbor area under baseline (i.e., existing) conditions. The baseline assessment is the first of a series of risk evaluations that will provide the basis for evaluating the need for and extent of remediation. It is based on existing conditions in New Bedford Harbor only; the potential natural decrease in contaminant mass and concentration in the harbor due to transport and degradation through time is not considered. Subsequent evaluations will examine the relative effectiveness of various remedial alternatives against current conditions using results of the numerical simulation model for PCBs.

EPA defines ecological risk resulting from toxic contaminants to include both direct risks to the growth, reproduction, or survival of the ecological receptor species, as well as the resource value of any species being reduced as a result of contaminant body burdens. Although both aspects of risk will be considered to some extent in this document, the former (direct) risk is the major concern of the assessment.

Ecological risks in New Bedford Harbor were determined by a mathematical evaluation and combination of two factors: (1) the degree of exposure to contaminants at the site, and (2) the ecotoxicity of PCBs and the three metals to aquatic organisms. Ecological risk was then quantified as the probability of impact to specific taxonomic groups representing the major ecotypes present in the harbor. Future evaluation of remedial alternatives via this method will require only repeating the exposure section of the assessment to reflect the new exposure conditions as determined by the numerical modeling results, and then using the previously derived (and unchanged) ecotoxicity calculations to determine new risk probabilities.

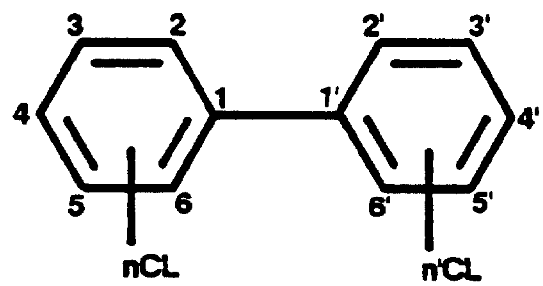
Following this strategy, this report consists of three sections. The first section is the exposure assessment, in which a representative subset of the organisms residing in the New Bedford Harbor area is identified, the routes of exposure are defined, and the degree of exposure is quantified. The second section, the ecotoxicity assessment, describes the acute and chronic toxic effects associated with PCB and metals exposure for each identified group. In addition, existing standards and criteria for PCBs and metals are discussed. The final section, the risk evaluation, combines the information presented in the two preceding sections to describe and quantify potential adverse effects on the New Bedford Harbor ecosystem resulting from the presence of these contaminants.

Both PCBs and metals are discussed in this report; however, PCBs were the primary focus of this study. Therefore, only the tables and figures for PCBs are included with the text. The tables and figures associated with the metals discussion are presented in Appendices A, B, and C.

The development of methodologies for determination of ecological risk is a relatively new and rapidly advancing field; the consensus among professionals concerning the most applicable methods at a particular site is limited. In addition, there are particular difficulties in determining risk due to PCBs in New Bedford Harbor because of the peculiar characteristics of PCBs as an environmental contaminant. PCBs are often treated as a single chemical or a small group of chemicals with similar properties; however, they actually consist of a group of 209 distinctly different chemical congeners. PCBs are relatively inert and, therefore, persistent compounds, with low vapor pressures, low water solubility, and high octanol/water partition coefficients. Although perhaps only half the potential congeners have actually been found to occur in the environment, they nonetheless consist of a diverse group of chemical species with widely varying physical, chemical, and biochemical properties.

In the manufacturing process, PCBs were formed by the addition of chlorine to the biphenyl molecule, and the number and types of PCB congeners formed in this process were not precisely determinable (Figure 1-3). Because PCBs were desirable primarily for their physical properties, which are largely related to the amount of chlorine substitution on the two rings, it was not necessary to know or control the exact congener mix; rather, only the percent of substituted chlorine in the final PCB mixture.

Most PCBs used in the U.S. were marketed as a mix of congeners under the name of Aroclor, a trade name of the Monsanto Company. Different Aroclors were designated by a four-digit code number (e.g., 1242 and 1254), with the last two digits signifying the amount of chlorine substitution as a weight percentage of the total mixture (e.g., Aroclor 1242 is 42 percent chlorine by weight). The sole exception to this numbering scheme is Aroclor 1016, which is approximately 41 percent chlorine. Aroclors 1016, 1242, and 1254 were most commonly used by the electrical component manufacturers in New Bedford. Because the desired properties of the Aroclors were determined by the overall amount of chlorination rather than the specific mix of congeners, it is probable that the actual congeners in a particular Aroclor varied among manufacturing batches. Reference Aroclors were subsequently established for analytical purposes; however, the relation of the reference Aroclors to the actual production batches is not clear.



(WHERE n AND n' MAY VARY FROM 0 TO 5)

FIGURE 1-3
CHEMICAL STRUCTURE OF A PCB MOLECULE
NEW BEDFORD, MASSACHUSETTS

After PCBs in the form of Aroclors are introduced into the environment, they begin to "weather," thereby changing and further complicating the problem of determining the actual mixture of components present. Lighter (i.e., less chlorinated) congeners are generally more volatile and soluble; therefore, they are (1) transported farther from the source before deposition, (2) less easily deposited into sediment, and (3) more easily mobilized and transported out of the original zone of deposition. More saturated congeners would demonstrate generally opposite behavior. In addition, differential rates of biochemical degradation, uptake, and depuration by biota, not easily related to level-of-chlorination but also determined by the actual pattern of chlorine substitution, would further serve to make the actual congener mix at any location different from the mixture originally released.

Although work is still ongoing to develop better analytical methods, it is possible to analyze environmental samples for many of the actual PCB congeners present; however, few congener-specific data are available because of the considerably greater analytical cost of the procedure. Most early studies reported PCBs as a "total" concentration or as the concentration of one or more Aroclors. Due to these problems, both methods produce less than completely satisfactory results. For the field sampling program conducted by Battelle Ocean Sciences (BOS) to produce calibration/validation data for the physical/chemical model (the source of much of the data used in this risk assessment), the analyses were reported in terms of "level-of-chlorination" homologs. This type of analysis provides valuable additional information, and because physical behavior determining fate and transport of PCBs is relatively similar for each homolog group, quantification (and subsequent numerical modeling) by homologs was deemed a reasonable cost-effective analytical goal for the modeling program. It was later decided to model only total PCBs, and the modeling program data were subsequently converted into total PCBs for risk assessment purposes by summing all homolog groups. Because the modeling and any remedial activities will be determined solely on the basis of total PCBs and, because of the lack of homolog-specific toxicity data, the risk assessment was conducted using total PCBs only.

The unique properties of PCBs and the problems with analysis described previously present considerable difficulties for determination of ecological (or public health) risk. Without analysis for specific congeners, it is not possible in most cases to know the actual congener

mix at a particular site, even if the exact congener composition of the PCBs introduced to the site were known, which is essentially never the case. Even if the mix of congeners were determined, the analysis would be valid only for the specific sample, and in an area such as New Bedford Harbor, the changing concentrations and mixture of congeners would present a complicated mosaic of spatial and temporal change. Therefore, the first step in conducting a risk assessment (i.e., determining the concentration of the contaminant(s) of interest at the specified site) is not possible for PCBs at the same level of detail as for other environmental contaminants.
Most analytical difficulties and uncertainties associated with determining PCB concentrations in the environment apply equally to any toxicological studies conducted with PCBs. A synthesis of the results of these studies is the second fundamental step in risk assessment and, because work to date has been conducted with contaminant concentrations reported as total PCBs or as one or more Aroclors, it is difficult to combine and use all data sources equally. Accordingly, various assumptions and simplifications were necessary at several points in the risk assessment so that the limited available data on PCB toxicity would not be unnecessarily reduced.

Recent work indicated substantial variability among congeners with regard to toxicity to aquatic organisms (Dill et al., 1982). Some toxicological properties are believed related to the configuration the two phenyl rings assume relative to each other which is, in turn, controlled by the position of the chlorines on the molecule. Fully ortho-substituted congeners do not assume a co-planar structure and are believed, in general, to be the least toxic. Conversely, non-ortho-substituted congeners are free to assume a co-planar configuration and are believed to be more toxic in general.

Site-specific water and sediment toxicity testing is perhaps the best solution to this problem; however, limited work has been conducted on New Bedford Harbor water and sediment. Although the availability of more data would have been valuable in that it would enable evaluation of the toxicity of the actual weathered PCB mixtures in New Bedford Harbor, it cannot prove that any effects measured are in fact due to the PCBs present rather than another contaminant. Therefore, both laboratory data on the toxicity of "pure" Aroclors and the limited data on actual toxicity of New Bedford Harbor environmental media must be used in combination to provide the "weight of evidence" for ecological risk.

aspects of modeling efforts by HydroQual, Inc. (Hydroqual) and Battelle Pacific Northwest Laboratories (PNL), various site investigation reports, the Greater New Bedford Health Effects Study, and the U.S. Army Corps of Engineers (USACE) Pilot Dredging Study and Wetlands Assessment. An extensive data base generated between 1981 and 1986 provides an accurate description of the current extent and level of contamination within most of the New Bedford Harbor area.

1.4.1 PCB Concentrations in Sediments

Data on distribution of PCBs in sediment and overlying waters of New Bedford Harbor and the Acushnet River Estuary were provided by PNL and BOS. For consistency with other aspects of the RI/FS process at the New Bedford Harbor site, the ecological risk assessment for PCBs was based primarily on a data set developed as the initial conditions for the physical/chemical transport model. Initial conditions were established by PNL using information on PCBs in the harbor obtained from three sources: (1) data collected by BOS (Duxbury, Massachusetts) specifically for the calibration and validation of the model; (2) a data base compiled by GCA Corporation (now Alliance Technologies Corporation [Alliance]) from various historical sources; and (3) a detailed survey of PCBs in the harbor conducted by NUS Corporation (NUS). These three data sets were subsequently combined into the central New Bedford Harbor data base by BOS. An additional intensive sampling of the Hot Spot provided the data used to establish concentrations in Hot Spot sediment.

1.4.1.1 BOS Calibration/Validation Data

From 1985 through 1986, BOS conducted four samplings of water, sediment, and biota in the Acushnet River Estuary, New Bedford Harbor, and adjacent areas of Buzzards Bay to provide data for calibration and validation of the physical/chemical transport model and food-chain model. Twenty-five stations were established and sampled on each of three surveys; the remaining survey was limited to eight stations and was conducted immediately following a storm event. Although the samples obtained during these surveys were collected and analyzed under rigorous quality control procedures, the data were intended for use primarily for model calibration/validation. The usefulness for determining patterns of contaminant distribution in New Bedford Harbor is limited by the relatively sparse spatial distribution.

The combination of these factors necessarily limits to some degree confidence in the accuracy of the risk probabilities for PCBs generated in this assessment, in the same way that confidence is decreased in using a statistical test to calculate probabilities when all assumptions for the test are not strictly satisfied. In some cases, it was possible to quantify the degree of uncertainty of some of the parameters and develop a quantitative estimate of overall uncertainty. For other issues, such as the question of congener-specific toxicity, it is not possible to approach the issue in a quantitative sense. However, because most toxicity studies have used congener mixtures, it is probable that a wide variety of toxicities is represented in both the test mixtures and the mixture occurring in New Bedford Harbor. The use of the risk probabilities in a relative sense (i.e., to compare the efficacy of different remedial alternatives against a no-action alternative) would have considerably greater validity, even if the absolute risk probabilities were questionable. It is this latter use that is important for the risk assessment.

Determination of risk due to heavy metals was not affected by the problems described previously for PCBs; however, other concerns became apparent during the analysis. Chief among these was the considerably smaller data set available for the three metals (particularly cadmium) and the probability that sampling for metals was concentrated in areas of suspected high concentrations, thereby biasing the data set. In addition, analysis of metals was deleted from the Battelle physical/chemical model and it was therefore not possible to work from the initial conditions established for each model cell, as was done for PCBs. This latter procedure would have largely corrected for the sampling bias. It was decided finally to use the available metals data exactly as provided thereby providing, to the extent that the data are biased toward higher concentrations, a more conservative estimate of risk.

1.4 PROGRAM DATA BASE

At most CERCLA sites, the ecological risk assessment would be based on findings of the RI report. However, because of the many studies conducted as part of the New Bedford Harbor project, numerous reports have been produced which obviate the need for a separate RI document. Therefore, this risk assessment is based primarily on the sampling data contained in the New Bedford Harbor data base,

1.4.1.2 Alliance Data Base

This previously compiled data base summarizing several of diverse field investigations in New Bedford Harbor represents an important source of data and was used extensively to set initial conditions for the model. The data base was originally constructed for EPA by Metcalf & Eddy, Inc., in 1983 and was transferred to Alliance in 1986. Alliance began to expand the data base and converted it to run under dBASE III, a personal computer data base management software package. This work was never completed, and the data base was subsequently provided to BOS for quality assurance checks and subsequent incorporation into the central New Bedford Harbor data base. The Alliance data base was provided to PNL by E.C. Jordan Co. (Jordan) as part of the data base PNL used to establish initial conditions for the physical/chemical transport model.

1.4.1.3 NUS Data Base

The NUS data base was provided to PNL in digital form by BOS. The data base was apparently complete and contained data for PCBs expressed as the concentrations of various Aroclors for samples obtained on a regular grid. The NUS data proved to be valuable because concentration data for the entire study area was provided. Data in the Alliance data base, for example, were concentrated at the Hot Spot and around various wastewater or combined sewer overflow discharges.

Details of the data selection, conversions, and manipulations conducted by PNL to establish the initial sediment PCB concentrations for the physical/chemical model will be discussed in the final modeling report currently in preparation (Battelle, 1990). In the remainder of this section, aspects of this process that are important for understanding this risk assessment are reviewed.

1.4.1.4 Selection of Data

Sediment PCB data from the BOS and NUS data sets were complete and easily interpretable, and were used as received. The Alliance data base contained a wide variety of contaminant measurements and included samples of air, water, wastewater, sediment, and biota from the general vicinity of New Bedford Harbor. In addition to data on PCBs and metals, the data base included data on water quality parameters and other organic and inorganic contaminants, most of which were irrelevant for establishing initial PCB concentrations for the modeling. PCB data were retrieved from the Alliance data base via a series of FORTRAN programs written by PNL.

1.4.1.5 Sample Depths

The BOS data base contained various combinations of samples taken at a number of different horizons in the sediment, gross (bulk) samples, and samples of different size fractions (i.e., sand, silt, and clay). Only gross (bulk) sediment samples from the upper stratum (5 centimeters) were retained for subsequent evaluation. The NUS data included samples taken from the upper stratum (6 inches), depths of 12 to 18 inches, and at specified greater depths. Only samples from the upper 6-inch stratum were retained.

Reflecting its multiple data sources, the Alliance data base included a wide variety of sampling horizons. The data records were divided into two categories: (1) surface samples obtained with a grab sampling device or collected as subsamples from the upper 8 inches of a sediment core; and (2) deep samples, for which any part of the subsample was taken from 8 inches or deeper below the sediment water interface. Only the surface samples were used in subsequent data analysis.

1.4.1.6 Data Conversions

The data sets used by PNL to establish the initial conditions for the modeling included PCB data in various forms. The most variation was encountered in the Alliance data base, in which PCBs were reported most commonly as Aroclors 1242, 1254, and 1242/1016, and non-specific PCBs. Some samples included data on level-of-chlorination homologs. The desired final measure, total PCBs, was obtained for each sample by summing the concentrations of all quantified Aroclors. Any samples reported on a wet-weight basis were converted to dry weight using an average water content of 55 percent.

PCB concentrations in the NUS data base were reported as Aroclor 1242, Aroclor 1248, or Aroclor 1254 in units of micrograms per kilogram, and assumed to be dry weight. Typically, only one or two Aroclor concentrations were reported for each sample. All reported Aroclor concentrations were summed and converted to units of micrograms per gram (ug/g), equivalent to ppm dry weight.

The BOS data base reported PCB concentrations by level-of-chlorination homolog in units of ug/g dry weight. These concentrations were summed to produce an estimate of total PCB concentration.

Values below specified detection limits occurred in all three data bases and were used in determining the initial conditions; values reported as zero were not used. Data reported below detection limits were assigned a value equal to approximately 0.1 times the specified detection limit of the analytical procedure and were placed in a separate file. When detection limits were not reported, concentrations of zero were assigned values of approximately 0.1 times the lowest reported value. These somewhat arbitrary assignments were necessary because the data were later log-transformed and values of zero would have been unacceptable.

1.4.1.7 Data Processing and Analysis

Standard univariate statistics were calculated by PNL for the raw and log-transformed data. The log-transformed data produced near-normal distributions around the mean value for each data set.

Contour plots of the surface sediment PCB concentrations were prepared at PNL and delivered to Jordan in November 1987. Initial PCB concentrations were calculated by PNL on a 100-by-100-foot grid and subsequently transferred to the larger i,j physical/chemical model grid by calculating an arithmetic average of all 100-foot grid data within each model grid element. The initial values for the i,j model grid, provided to Jordan by PNL in April 1989, were used for all subsequent analyses conducted for the ecological risk assessment, with one modification at the Hot Spot. Following the final assignment of initial conditions for the model, USACE funded an additional intensive survey of PCB concentrations in the Hot Spot. Three model grid cell concentrations were changed from initial condition assignments to reflect the updated information.

1.4.2 <u>PCB Water Concentrations</u>

PCB concentrations in the water column for the risk assessment were also based on values used for the physical/chemical transport model. However, unlike sediment concentrations, the use of initial conditions is not appropriate because preliminary model runs indicated that concentrations in the water column are determined largely by the assigned sediment concentrations following a brief "spin-up" period of approximately 90 days simulation. Accordingly, PNL did not determine initial conditions for the water column in a manner similar to that previously described for sediment; rather, it assigned initial conditions generally consistent with the field data and then allowed the model to produce its own "starting conditions" based on the assigned sediment concentrations. These starting conditions in the water column were averaged vertically for each cell in the i,j grid and provided to Jordan with the initial sediment conditions.

1.4.3 Metals Concentrations

Because metals were not included in the Battelle physical/chemical modeling effort, it was not possible to use model initial conditions for the calculation of exposure estimates at the New Bedford Harbor site. Metals data were obtained from the program data base maintained by BOS. All data for the three metals in water and sediment were requested and received via magnetic disk. Data characterized as "rejected" in the data validation were removed from the data set and not used in the risk assessment. The data set contained numerous "non-detects," which were entered into the analysis as half the lowest reported concentration for the particular metal. All remaining data were used as received.

1.5 OVERVIEW OF METHOD FOR THE ECOLOGICAL RISK ASSESSMENT

A joint probability model was used in the risk assessment to quantitatively evaluate potential impacts to New Bedford Harbor biota for each contaminant. The basic components of the model are two probability distributions, one representing the expected distribution of contaminant levels in the environment, and the second representing the probability distribution of some benchmark concentration for a particular group of potential receptors over a range of contaminant levels. The joint probability model is used to determine the likelihood that a typical species (which displays a particular biological effect at the benchmark concentration) will encounter an environmental concentration sufficient to elicit the particular effect.

In Subsection 2.1.2, development of the expected distribution of environmental levels is discussed. These distributions are termed expected environmental concentration (EEC) probability curves. The development of the probability density function that relates contaminant concentration to a biological benchmark is discussed in Subsection 3.2. Finally, the joint probability model is used to determine quantitative risk estimates in Section 4.0.

2.0 EXPOSURE ASSESSMENT

The environmental exposure assessment was performed to identify representative organisms within New Bedford Harbor that may be exposed to PCBs and metals. The assessment included identification of ecological receptors and exposure routes, with the goal of selecting a subset of species to represent the wide variety of potential aquatic receptors at the site. These species were used to identify the principal routes of exposure and describe contaminant exposure within the New Bedford Harbor area.

For the purposes of accumulating results at various (simulated) points in time, the Battelle transport model divides the estuary and harbor into the following five zones, based in part on natural and manmade structures and on the initial contaminant concentrations detected in the sediment (Figure 2-1):

- Zone 1: the area between the Wood Street Bridge and the southern boundary of the Hot Spot
- Zone 2: from the southern boundary of the Hot Spot to the Coggeshall Street Bridge
- Zone 3: the area between the Coggeshall Street Bridge and Popes Island (State Route 6 Bridge)
- Zone 4: the area between Popes Island (State Route 6 Bridge) and the Hurricane Barrier
- Zone 5: from the Hurricane Barrier out to the limit of the modeling grid, roughly delineated by the line from Ricketsons Point to Wilbur Point

Different systems of dividing New Bedford Harbor into zones have been used at various times for specific purposes. The zone definition used in this report for the purpose of the ecological risk assessment is identical to the zonation being used for the physical/chemical transport modeling. The risk assessment is based primarily on both the input to and output from the model, and use of the same zones simplified inclusion of the data from modeling runs. Therefore, slightly different divisions of the harbor were used for the HydroQual food-chain model, the public health risk assessment, and the draft ecological risk assessment.

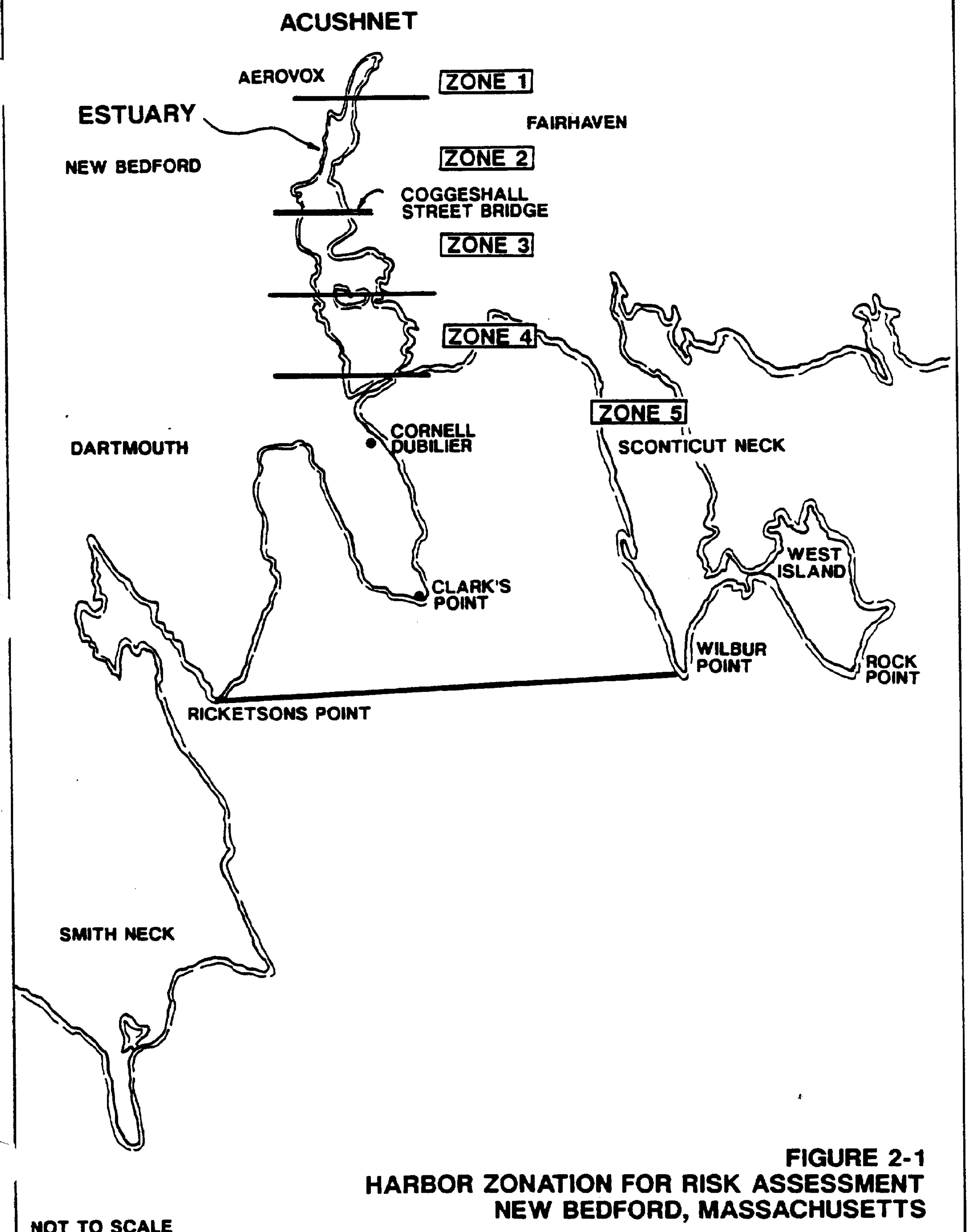


FIGURE 2-1
HARBOR ZONATION FOR RISK ASSESSMENT
NEW BEDFORD, MASSACHUSETTS

Although all these divisions correspond in some areas to the various fishery closure zones, none is exactly the same.

2.1 RECEPTOR IDENTIFICATION

2.1.1 Exposed Species Analysis

Many organisms in New Bedford Harbor are potentially at risk as a result of exposure to PCBs and heavy metals. The four primary routes of exposure include (1) direct contact with the water in the water column, (2) direct contact with or ingestion of sediment, (3) direct contact with sediment pore water, and (4) ingestion of contaminated food. The route of exposure can also be defined by the method of obtaining food (e.g., herbivore, carnivore, suspension feeder, deposit feeder, and scavenger). To describe how aquatic organisms may be exposed to contaminants at the New Bedford Harbor site, a representative subset of the species known to inhabit this area was identified. The basis of the selection was defined by the possible routes of exposure for the organisms in question.

To evaluate the level of effects due to exposure and for risk characterization, the organisms in New Bedford Harbor were separated into ecotypes, which also correspond to taxonomic groups. Five groups of organisms, corresponding to the major aquatic organisms present in the harbor and also representative of the range of exposure routes, were developed: marine fish, crustaceans, mollusks, polychaetes, and algae. The rationale for these groupings and typical representative species for each in New Bedford Harbor are presented in Section 3.0. Lack of toxicological data for marine polychaetes precluded separate analysis of potential contaminant effects on this group. However, these organisms are considered relatively insensitive to organic contamination in sediment and are widely used for bioaccumulation studies for this reason. In the determination of risk in Section 4.0, it is assumed that a typical polycheate would be no more sensitive than a typical mollusk, and the benchmark distribution for mollusks will be used conservatively to assess risk to polychaetes as well.

Although most organisms can be exposed to environmental contaminants via all media, for purposes of assessing exposure in this risk assessment, the various habitat locations (i.e., benthic or pelagic), lifestages (i.e., egg, larvae, and adult), and feeding method (e.g., filter feeder, deposit feeder, or carnivore) of typical members of each group were used to define the primary routes of exposure for the group. Based on habitat, direct contact with dissolved or particulate contaminants in the water column was considered the primary route of exposure for pelagic fish, bivalves, and plankton. An important secondary route of exposure for most species is consumption of biota that have bioaccumulated contaminants. For benthic infaunal invertebrates, it was determined that direct contact with and ingestion of contaminated sediment and food organisms were the primary routes of exposure. Direct contact with the water column was determined to be a secondary route of exposure, although it can also be the primary exposure route for planktonic lifestages of infaunal adults.

2.1.2 Species of Concern

Species of concern inhabiting the New Bedford Harbor area were identified based on the biological surveys conducted by IEP, Inc., for USACE (USACE, 1988b); Sanford Ecological Services for USACE (USACE, 1986); Camp, Dresser and McKee (Camp, Dresser and McKee, 1979); and historical data reported in Bigelow and Schroeder (Bigelow and Schroeder, 1953).

A subset of receptor species was selected from these data based on the following criteria: distribution within the study area, trophic level (i.e., producer, primary, secondary, or tertiary consumer); commercial and/or recreational use; and availability of biological and ecological information.

Criteria such as habitat location, trophic level, and reproductive potential are important factors that may influence the ways in which each species may be exposed to contaminants in the New Bedford Harbor area and the potential effects of contaminant exposure. The commercial and/or recreational value of a resource species is a key factor for species selection because the loss and limitation of use of such species may have economic significance.

Twenty-eight species of various trophic levels and habitat types representing the five taxonomic groups of aquatic organisms discussed previously (i.e., finfish, crustaceans, mollusks, annelids, and plankton) were selected as typical aquatic receptors for the New Bedford Harbor site. Distribution of these species within the Acushnet River/Buzzards Bay area is shown in Table 2-1.

E 2-1

DISTRIBUTION OF THE 28 SELECTED IES OF CONCERN IN NEW BEDFORD HARBOR

NEW BEDFORD HARBOR

ALL ZONES	ZONE 1 (AREA 1)	ZONE 2 (AREA 1)	ZONE 3 (AREA 1)	ZONE 4 (AREA 1)	ZONE 5 (AREA 2)
Fish					
Herring	American Eel	American Eel	Scup	Scup	Scup
Flounder			Tautog	Tautog	Tautog
Silverside			American Eel	Mackeral	Mackeral
Mummichog					
Crustaceans					
	Isopod	Blue Crab	Blue Crab	Green Crab	Lobster
		Fiddler Crab	Green Crab	Lobster	Amphipod
		Green Crab	Lobster	Grass Shrimp	
		Amphipod	Fiddler Crab		
			Amphipod		
			Grass Shrimp		
Mollusks					
Quahog	Mud Nasa	Mud Nasa	Blue Mussel	Blue Mussel	Quahog
Ribbed Mussel	Soft-shell Clam	Soft-shell Clam	Slipper Shell	Slipper Shell	
		Blue Mussel	Bay Scallop	Eastern Oyster	
		Quahog	Soft-shell Clam	Quahog	
			Eastern Oyster		
			Quahog		
Plankton					
Diatoms		Copepod	Copepod	Copepod	Copepod
Annelids					
Clam Worm					
Mud Worm					
Thread Worm					

NOTE:

Zones correspond to Figure 2-1; areas correspond to Figure 1-2.

3.88.80
0023.0.0

2.2 EXPOSURE LEVELS FOR RECEPTORS

2.2.1 Introduction

The amount of contaminant exposure experienced by an aquatic organism is a function of the type(s) of contaminated media to which the organism is exposed, contaminant concentrations in the media, and the mechanisms by which contaminants are taken up from each medium. Each factor was considered and, to the extent possible, quantified, in determining exposure levels for the five organism groups used for the risk assessment.

PCB contamination in New Bedford Harbor has been documented in all environmental media (i.e., water, sediment, and biota) throughout the harbor; however, it varies considerably in concentration, generally decreasing with distance from the Hot Spot in the Upper Estuary. Metals contamination is similarly ubiquitous; however, the area of highest metals concentrations is found in Zone 3 between the Coggeshall Street and Popes Island bridges. Organisms residing in New Bedford Harbor for all or part of their lives may be exposed to these contaminants as a result of direct contact with and/or ingestion of contaminated food, water, and sediment. Migration from the harbor of prey species with elevated PCB and metals tissue burdens expands the potential area of exposure for predators. Uptake of contaminants from water, sediment, or food into the tissues of organisms ultimately occurs by either passive diffusion, active transport, or facilitated transport across the membranes of the gills, gastrointestinal lining, mouth lining, and body wall (Swartz and Lee, 1980).

Terms such as bioconcentration and bioaccumulation relate to the source and specific outcomes of exposure to contaminants. Bioconcentration refers to the net uptake of dissolved chemicals into an organism from water. Another directly related term, bioconcentration factor (BCF), is the ratio of concentration found in the tissue of an organism to the concentration in the water to which the organism was exposed (Schimmel and Garnas, 1985). The term bioaccumulation refers to the net uptake of a contaminant by an organism from all sources, including ingestion of and/or contact with water, food, and sediment (Menzer and Nelson, 1986). Biomagnification is generally used to refer to the concentration of a contaminant between trophic levels in a food chain.

2.2.2 Methods

PCB concentrations in the water column (i.e., dissolved concentration), pore water, and sediment developed as initial conditions for the modeling program were the primary sources of exposure data for the ecological risk assessment. The source and development of the initial condition concentrations are discussed in Subsection 1.4. For the Upper Estuary Hot Spot, the initial conditions data were supplemented with concentrations obtained from the USACE data set for this area (USACE, 1988c).

The modeling program PCB data were provided as total bed sediment concentrations and vertically averaged water column concentrations for each element in the i,j grid used for the physical/chemical model. Each data point was weighted equally for subsequent analysis; however, there is some variation in the size and, therefore, the amount of the harbor represented by each model grid element. Hot Spot concentrations, assumed to represent the range of concentrations present in the Hot Spot, were also weighted equally.

All data were log-transformed and assigned to one of six groups representing the Hot Spot and each of the five zones of the harbor discussed previously (see Figure 2-1). Simple descriptive statistics (mean and variance) were calculated for each zone and used to generate an EEC probability function for each zone. EECs are cumulative frequency distributions that quantify the likelihood that the actual environmental concentration at any location in a zone will be equal to or less than a particular value.

Because the joint probability model used to estimate risks in Section 4.0 presumes that the EEC and the effects distributions are normally distributed, the log-transformed PCB concentration data for each harbor zone were examined for deviations from normality using the Kolmogorov-Smirnov test (i.e., a=0.05). In most cases, results indicated that the transformed concentration data are not normally distributed. No other transformations were attempted to rectify this problem, because the toxicological data used in development of effects curves are log-normally distributed, and the same scales must be used for both the EEC and effects distributions to determine a joint probability risk estimate. Also, examination of the moment statistics for EEC distributions indicated that the major reason distributions are not normally distributed is due to leptokurtosis rather than skewness. In contrast with skewed distributions, the distributions are symmetrical around the mean value, and deviations from normality are less problematical.

Data reduction and analysis for metals was conducted following procedures essentially similar to those described previously for PCBs, the primary difference being that raw data from the program data base maintained by BOS were used in place of initial conditions for the physical/chemical model.

2.2.3 Exposure to Water Column Contamination

2.2.3.1 Species and Mechanisms

Organisms exposed to contaminants primarily via the water column include pelagic or planktonic species that live suspended or swimming in the water column, and demersal finfish that may have some contact with the bottom but receive most exposure from the water. Representative pelagic and demersal fish found in the New Bedford Harbor area include winter flounder (Pseudopleuronectes americanus), bluefish (Pomatomus saltatrix), blueback herring (Alosa aestivalis), and Atlantic silverside (Menidia menidia).

Phytoplankton and zooplankton are also exposed nearly exclusively via contaminants in the water column. Although effects on holozooplankton and phytoplankton are usually not of direct concern, their importance for higher trophic levels can be significant. Representative plankton in New Bedford Harbor include the copepods (Acartia tonsa) and two diatoms (Rhizosolenia alata and Skeletonema costatum). The opossum shrimp (Neomysis americana) is generally considered epibenthic rather than planktonic; however, for the purposes of the risk assessment, its behavior is sufficiently similar to planktonic organisms that it can be considered part of the planktonic group.

Bivalve mollusks, although seemingly species that would be exposed via sediment, are primarily exposed to waterborne contaminants due to the filtering of large amounts of water to extract food. In addition, bivalve mollusks have planktonic larval stages that are also exposed to contaminants in the water column. Representative bivalves in New Bedford Harbor include the Atlantic ribbed mussel (Geukensia demissa), the blue mussel (Mytilus edulis), the Atlantic bay scallop (Aequipecten irradians), and the Eastern oyster (Crassostrea virginica).

For all these organisms, the epithelial tissue of the gills is usually the primary site of contaminant uptake because of its structure and function. Uptake of contaminants from water can also occur across the linings of the mouth and gastrointestinal tract, the sensory organs, and even the viscera if they are perfused with water, as in some mollusks. Waterborne contaminants can also become adsorbed onto exposed surfaces such as the skin, where they may disrupt the function of some tissues but do not generally contribute to systemic toxicity.

2.2.3.2 PCB Exposure Concentrations in Water

Exposure levels in the water column are for the disssolved concentrations of PCBs. The dissolved component in the water column, as opposed to total concentrations, was used because most data about toxicological effects of PCBs on organisms are based on dissolved concentrations. Therefore, assessing the impact of dissolved concentrations of the contaminant more directly relates to the toxicological data. The concentration is the average for the entire water column. The mean, standard deviation, and variance for each zone are listed in Table 2-2. Cumulative probability plots for the water column exposure levels, presented in Figure 2-2, are based on a random sample of 100 data points from distributions with the calculated parameters (see Table 2-2). As shown in Table 2-2, the mean water column PCB levels decrease with increasing distance from the Hot Spot in Zone 1. Despite the large difference in the number of grid elements for the various zones, the variances associated with the different zones are similar. Mean values for Zone 1 and the Hot Spot are 2.55 and 3.10 micrograms per liter (ug/L), respectively, decreasing to 0.02 ug/L in Zone 5.

Because of the similarity in the variances associated with the environmental concentration data, the shape of the resulting EEC curves are similar, differing mainly in location along the PCB concentration axis (see Figure 2-2).

2.2.3.3 Metals Exposure Concentrations in Water

The exposure levels in the water column for all metals are for the dissolved concentrations of the metals. As in the case of PCBs, the dissolved component was used rather than the total concentration because most of the data about toxicological effects of metals are based on dissolved concentrations. The geometric mean, standard deviation, and variance for each zone are in Appendix A; that is, Table A-1 for copper, Table A-2 for cadmium, and Table A-3 for lead. The cumulative EEC probability plots for all zones for copper, cadmium, and lead are presented in Figures A-1, A-2, and A-3, respectively.

There is little indication of any relationship between the concentrations of copper and cadmium, and distance from

TABLE 2-2
EXPECTED EXPOSURE CONCENTRATIONS FOR PCBS (1)

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

HARBOR ZONE	MEAN (ug/l)	TRANSFORMED VALUES (2)		
		MEAN	STANDARD DEVIATION	VARIANCE
Hot Spot, Water Column	3.097	0.491	0.128	0.016
1. Water Column	2.559	0.408	0.139	0.019
2. Water Column	1.074	0.031	0.272	0.074
3. Water Column	0.157	-0.804	0.250	0.063
4. Water Column	0.065	-1.185	0.099	0.010
5. Water Column	0.023	-1.639	0.255	0.065
Hot Spot, Pore Water	73.114	1.864	0.642	0.767
1. Pore Water	38.282	1.583	0.302	0.091
2. Pore Water	4.406	0.644	0.954	0.910
3. Pore Water	0.277	-0.558	0.393	0.154
4. Pore Water	0.075	-1.125	0.708	0.502
5. Pore Water	1.000	-1.320	0.551	0.303

NOTES:

1. All data developed using initial conditions for Battelle numerical model. Expected pore water concentrations derived from initial sediment concentrations times model mass-transfer coefficient.

2. Log (base 10) transformed values, with standard deviations and variances.

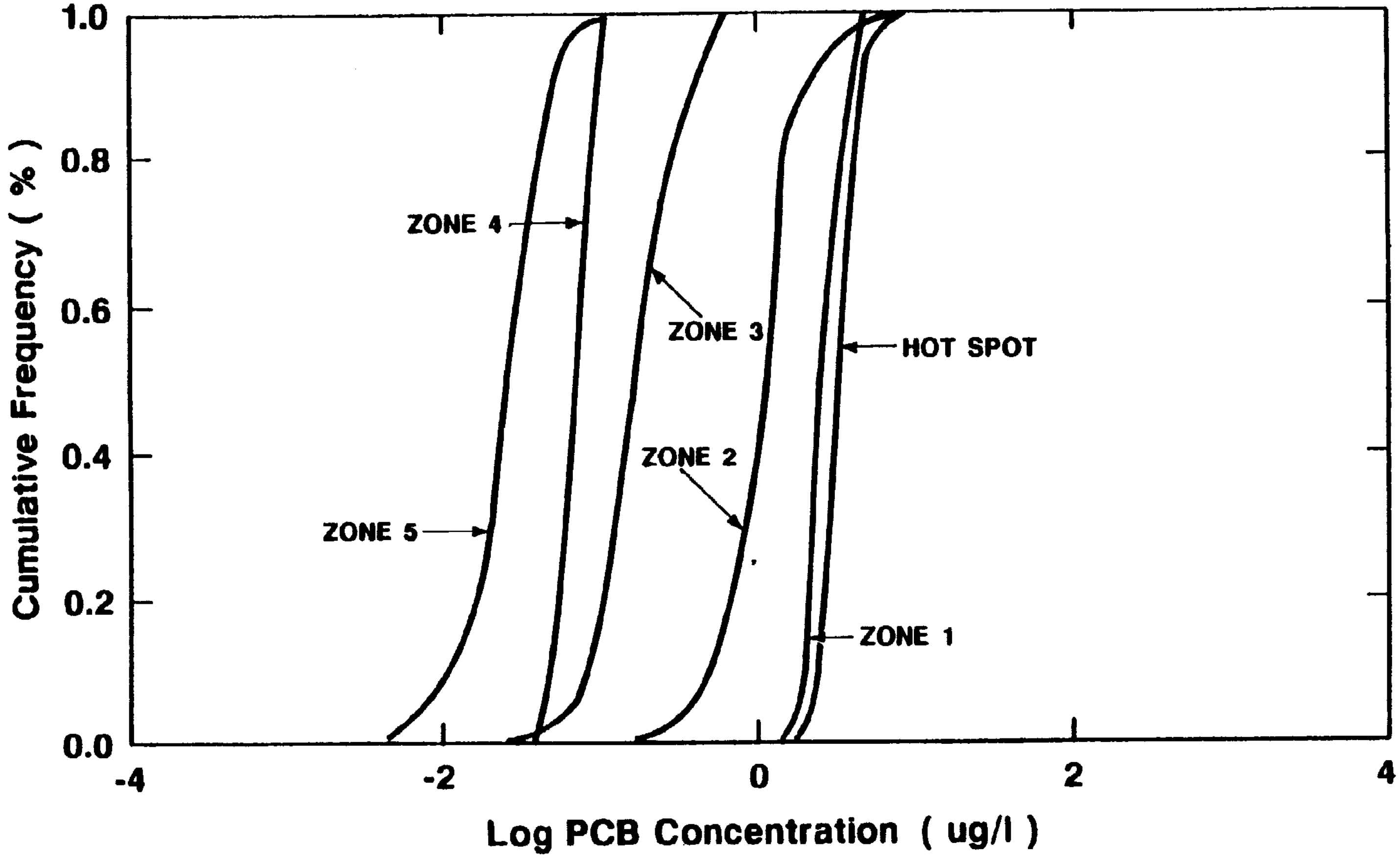


FIGURE 2-2
EECs BY ZONE FOR PCBs, WATER COLUMN
NEW BEDFORD, MASSACHUSETTS

the Upper Estuary, as was found with PCBs. However, there is a noticeable decrease in lead concentrations with increasing distance from Zone 1; within zones, lead concentrations were more variable than copper and cadmium concentrations.

2.2.4 Exposure to Sediment Contamination

2.2.4.1 Species and Mechanisms

Direct contact with and ingestion of contaminated sediment and its associated pore water are the primary routes of exposure for benthic infauna that live in close association with or are buried in the sediment. Exposure of epifaunal benthic organisms is more difficult to quantify because they are exposed to both sediment and the overlying water; for these species, exposure primarily to sediment can be used as a conservative worst case. Typical benthic invertebrates in New Bedford Harbor include the American lobster (Homarus americanus), amphipod (Ampelisca vadorum), tubificid worm (Tubificoides sp.), slipper shell (Crepidula fornicata), and mud snail (Ilyanassa obsoleta).

In the environment, sediment usually provides the most concentrated pool of contaminants, as evidenced at the New Bedford Harbor site (Larsson, 1985). For most of the contaminated sediment in the harbor, PCBs and metals are continually being released into the interstitial or pore water, from which uptake by benthic organisms occurs. Resuspension of sediment also increases total contaminant concentrations in the water column, but these particulate-bound contaminants are not directly available for uptake as are the dissolved-phase contaminants.

Sediment-bound contaminants are also taken up directly from the sediment by aquatic organisms (O'Donnel et al., 1985). Deposit-feeding organisms that feed by ingesting sediment also ingest any contaminants bound to the sediment. Contaminants strongly bound to sediment are less likely to desorb from sediment particles, and are absorbed in the gut less than the more weakly bound contaminants. Uptake may also occur as a result of equilibrium partitioning of contaminants between the body surfaces of the organism and surface coatings of the sediment (Swartz and Lee, 1980).

Although these various modes of uptake have all been documented, a quantitative assessment of risk incorporating all the mechanisms is not possible because of the lack of sufficient relevant toxicological data. Therefore, risk for benthic organisms was defined as risk due to exposure to contaminants dissolved in pore water. By assessing risk in this form, it is possible to draw on the body of toxicological data that has largely been developed using dissolved contaminants.

2.2.4.2 PCB Exposure Concentrations in Sediment Pore Water

PCB concentrations in pore water were calculated from the initial conditions sediment concentration data for the physical/chemical model via partition coefficients (K_d). Because of the properties of PCBs discussed in Subsection 1.3, partitioning is a complex phenomenon that varies over several orders of magnitude according to specific PCB congeners. Because the PCBs present in New Bedford Harbor represent a mixture of congeners, no single K_d can fully describe the partitioning that is occurring.

Values for site-specific apparent K_d in New Bedford Harbor are available from experiments conducted by BOS as part of the modeling program, and from the literature (Brownawell and Farrington, 1986). The K_ds ultimately selected were numerically equivalent to the mass transfer K_ds used in the physical/chemical model to approximate diffusion of dissolved PCBs from bed sediment, and are generally comparable to K_ds determined empirically by BOS, and consistent with the range of values reported in other studies (Brownawell and Farrington, 1986; and Pavlou and Dexter, 1979).

For areas above the Coggeshall Street Bridge (i.e., Zones 1 and 2), the K_d used was $5x10^5$; below the Coggeshall Street Bridge (i.e., Zones 3, 4, and 5), the K_d used was $2x10^5$. The K_ds were applied to the original data and the results log-transformed. Descriptive statistics were calculated as described for water concentrations, and the results are summarized in Table 2-2. As with the water column data, estimated pore-water PCB concentrations are highest in the Hot Spot, decreasing with distance from this area. Mean values for Zone 1 and the Hot Spot are 38.28 and 73.11 ug/L, respectively, decreasing to 0.05 ug/L in Zone 5. As was the case with data for water column PCB levels, variances associated with estimated pore water levels for the different zones are comparable, resulting in similarly shaped EEC curves (Figure 2-3).

2.2.4.3 Metals Exposure Concentrations in Sediment Pore Water

Exposure levels for metals in the pore water were calculated from the sediment concentrations via K_ds.

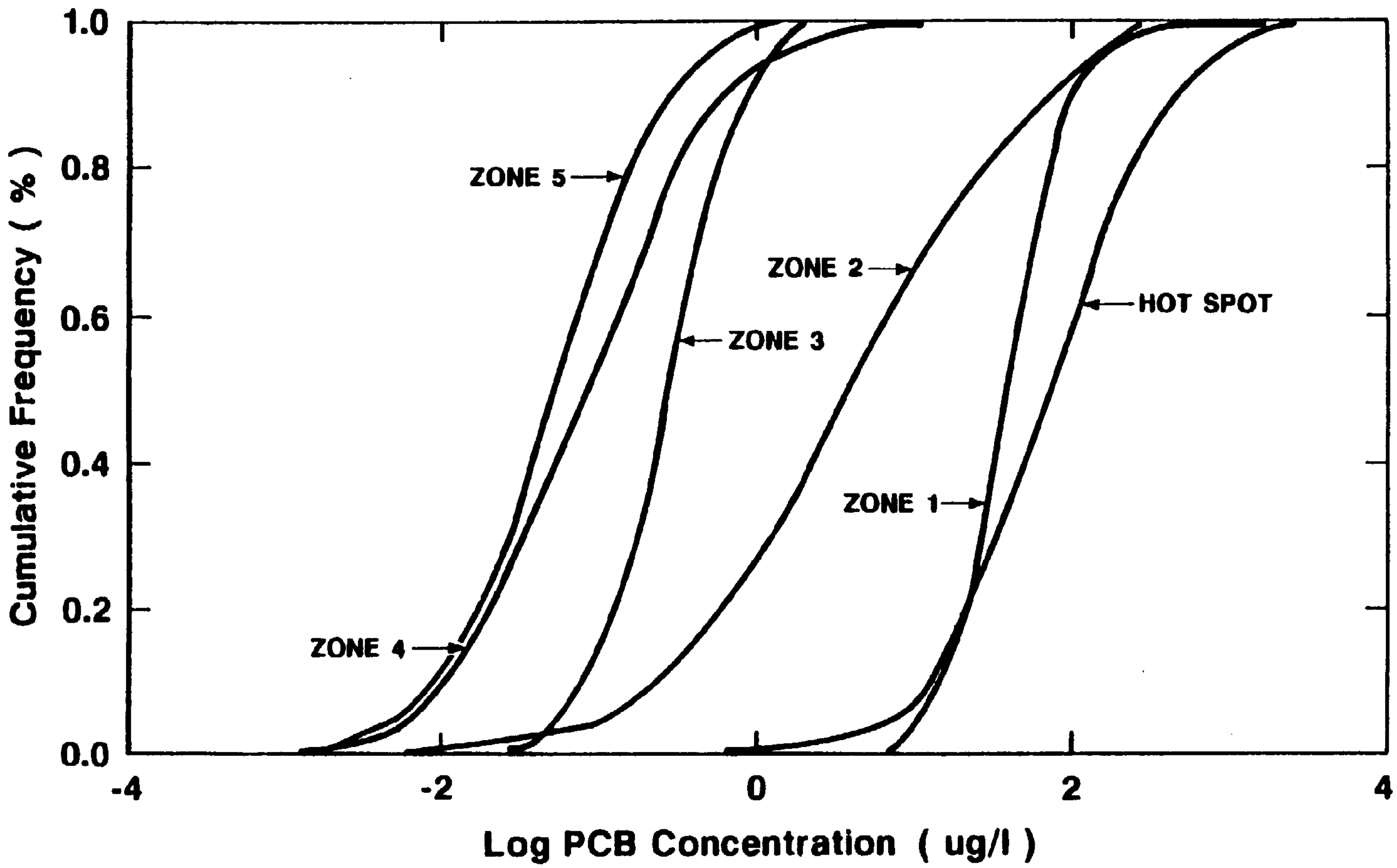


FIGURE 2-3
EECs BY ZONE FOR PCBs, PORE WATER
NEW BEDFORD, MASSACHUSETTS

The K_ds used were based on field measurements made throughout the New Bedford Harbor site, provided by Damian Shea from BOS (unpublished masters thesis). The K_ds used were $8x10^5$ for copper, $4x10^4$ for cadmium, and $2x10^5$ for lead.

The mean, standard deviation, and variance for each zone are presented in Table A-1 for copper, Table A-2 for cadmium, and Table A-3 for lead. The cumulative EEC probability plots for all zones for copper, cadmium, and lead are presented in Figures A-4, A-5, and A-6, respectively.

Calculated pore water concentrations of copper and cadmium were the lowest in Zone 5 and the highest in Zones 1 and 3 (Figures A-4 and A-5). Lead concentrations in the pore water were the lowest in Zone 4 and the highest in Zones 1 and 3. For all metals, the highest variance was associated with Zone 2. As with the water column concentrations, a decrease in concentrations with increasing distance from the PCB Hot Spot is not as well defined as for PCB concentrations, although a weak trend can be observed.

2.2.5 Exposure to Contaminated Food

Allotrophic organisms in New Bedford Harbor are exposed to PCBs and metals via ingestion of contaminated food. Lipophilic organic compounds (e.g., PCBs) transfer efficiently across the gut membranes because of the relatively long contact time between food and membranes. The consumption of contaminated food is of concern if dietary intake directly results in toxicity, and/or if the chemical is subject to food-chain transfer resulting in tissue burdens that may potentially be toxic.

A food-chain model is being developed for the New Bedford Harbor site by HydroQual. The transfer and fate of PCBs and metals are being assessed with the model for two different food chains, culminating in American lobster (*Homarus americanus*) and winter flounder (*Pseudopleuronectes americanus*), respectively (Figures 2-4 and 2-5).

The HydroQual model consists of a series of differential equations that numerically simulate the various processes that determine the residue value, or amount of a contaminant that remains in the tissues of the organism over time. Processes simulated in the model include surface sorption, transfer across the gills, ingestion of

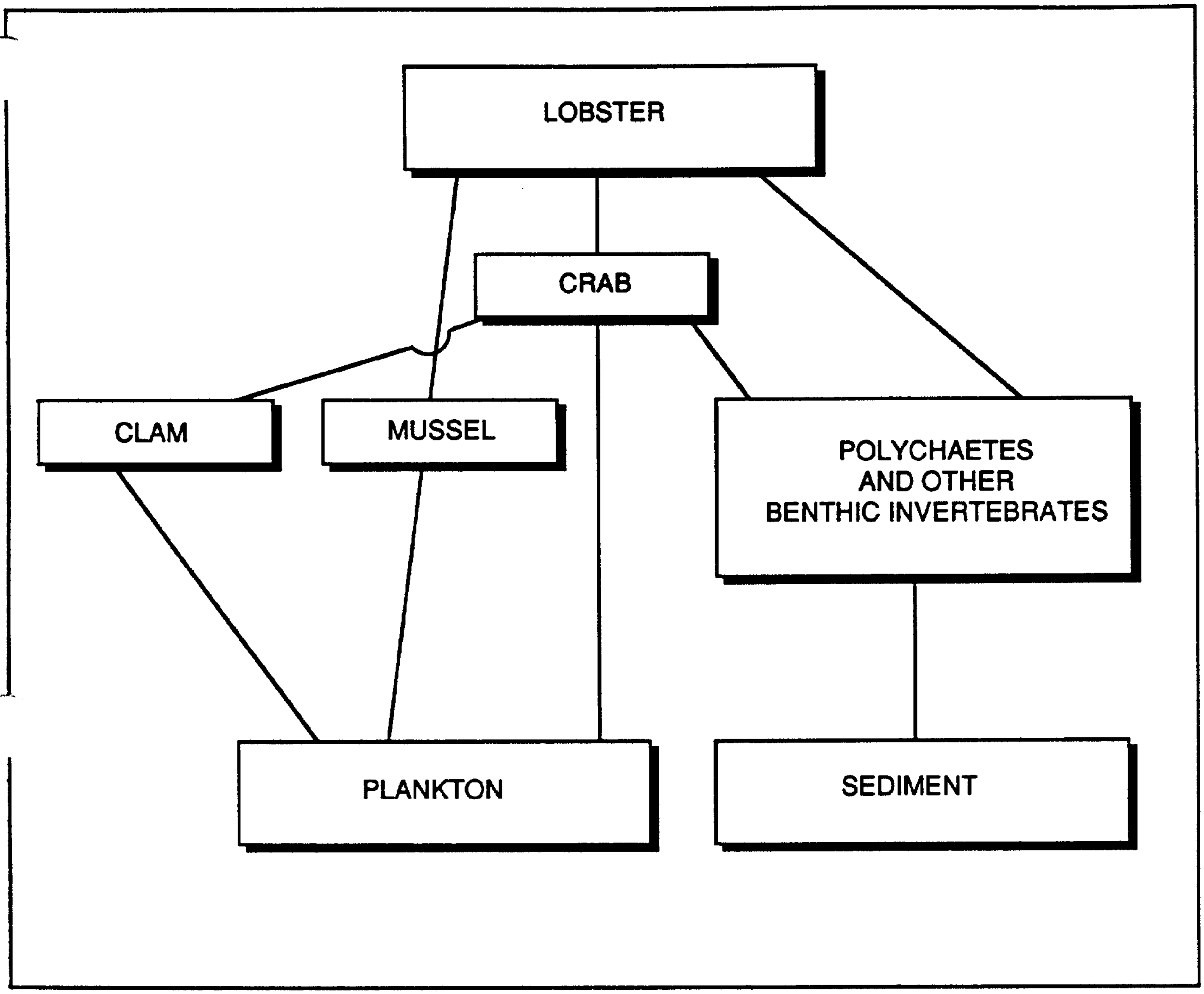


FIGURE 2- 4
LOBSTER FOOD CHAIN
NEW BEDFORD, MASSACHUSETTS

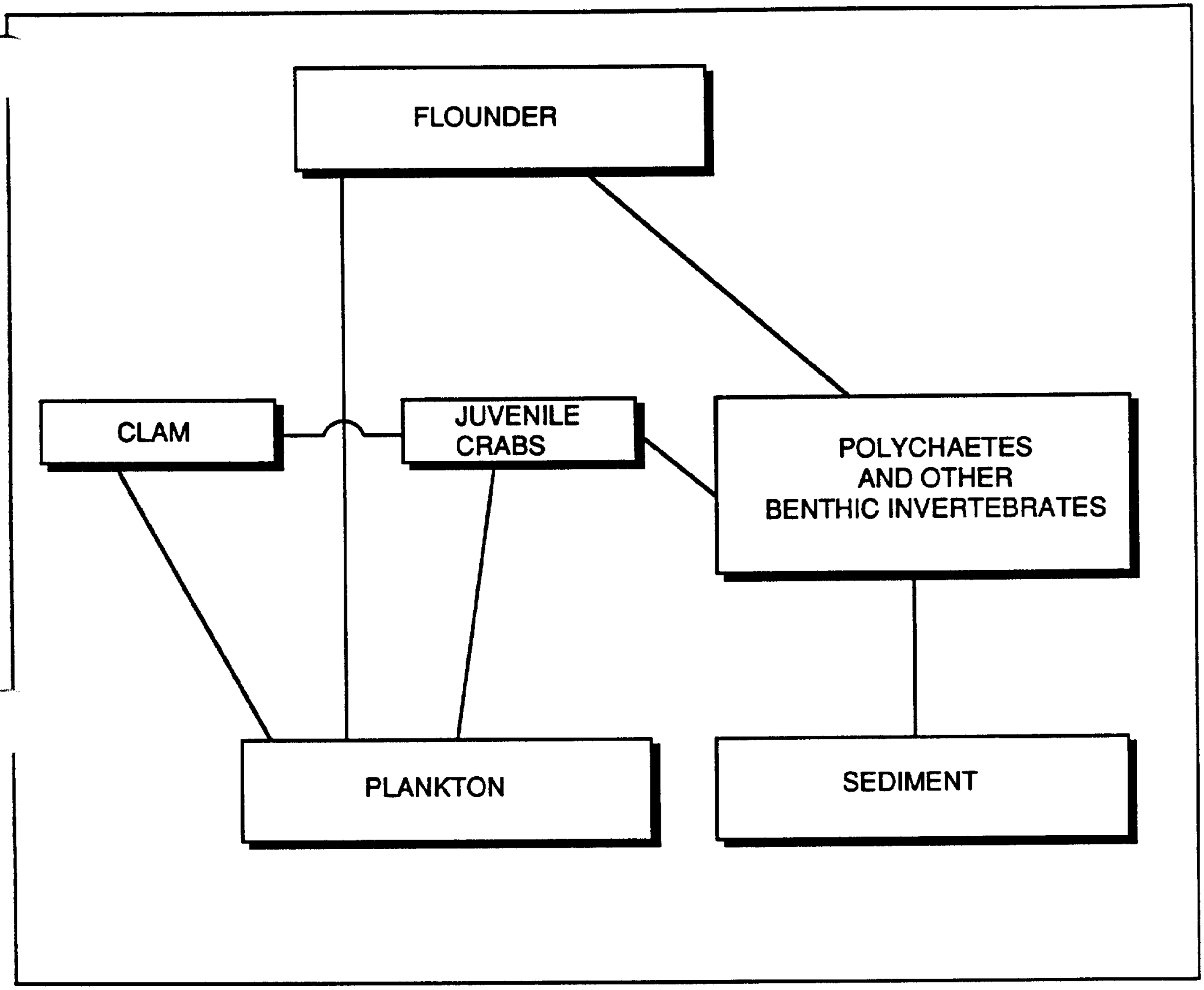


FIGURE 2- 5
FLOUNDER FOOD CHAIN
NEW BEDFORD, MASSACHUSETTS

contaminated food, desorption, metabolism, excretion, and growth. These processes are regulated by the physical/chemical characteristics of PCBs and by the physiological processes of the biota.

The food-chain model is designed to predict residue concentrations in species consumed by humans; therefore, it is a component of the public health risk assessment, as well as the ecological risk assessment. Because there are relatively few data available on the effect of residue values on aquatic biota, it is not possible to use the model results directly in the ecological risk assessment. The model does not include provisions for modifying any of the physiological processes as the organisms become stressed due to increasing body burdens of contaminants. However, it is necessary to consider toxic effects due to residue values as part of the risk assessment (see Section 4.0).

Also of importance for the risk assessment is the observation, based on calibration and validation of the food-chain model, that consumption of PCB-contaminated food may account for the majority (up to 95 percent) of PCB residue concentrations in aquatic species in New Bedford Harbor, although other investigators consider this figure unreasonably high for all but top predators (Hansen, 1990). Therefore, although there are insufficient data to evaluate this pathway quantitatively, it must be considered in some way if the risk assessment is to reflect actual effects on aquatic biota in New Bedford Harbor. This aspect of ecological risk is discussed in Section 4.0.

The mean levels (and ranges) of PCB tissue concentration found in organisms in the New Bedford Harbor area are summarized in Table 2-3, which is based on levels found in samples collected during the Battelle cruises of 1984, 1985, and 1986. These data indicate that PCB tissue residue concentrations are correlated with the levels of PCBs found in the New Bedford Harbor sediment and water column. For the six species comprising varied trophic levels and habitat preferences, highest tissue burdens were found in organisms collected from the inner harbor; levels decreased in successive areas in the outer harbor. The highest tissue levels were observed in polychaete worms, which are in direct and continuous contact with highly contaminated sediment. Winter flounder (Pseudopleuronectes americanus) also had relatively high whole-body tissue levels, perhaps reflecting its position in the marine food web and its habit of lying partially covered by bottom sediments.

TABLE 2-3
WHOLE-BODY CONCENTRATIONS OF TOTAL PCBS (PPM) IN ORGANISMS
COLLECTED FROM NEW BEDFORD HARBOR

NEW BEDFORD HARBOR

SPECIES	LOCATION[1] AREA 1	AREA 2	AREA 3	AREA 4
American Lobster				
Minimum	---	0.195	0.042	0.017
Mean	1.131[2]	0.568	0.213	0.064
Maximum	---	1.235	0.351	0.176
Winter Flounder				
Minimum	3.138	0.926	0.515	0.123
Mean	7.992	2.853	2.138	0.777
Maximum	20.230	8.067	6.349	2.616
Mussel				
Minimum	1.467	1.461	0.254	0.008
Mean	2.262	3.874	0.266	0.023
Maximum	2.962	6.204	0.278	0.039
Quahog				
Minimum	0.200	0.010	0.026	0.200
Mean	5.300	1.777	1.200	0.300
Maximum	2.121	1.182	0.478	0.137
Green Crab				
Minimum	0.071	0.067	0.624	0.020
Mean	0.398	0.184	0.976	0.048
Maximum	0.725	0.301	1.329	0.077
Polychaetes				
Minimum	---	---	0.096	0.182
Mean	12.972[2]	1.654[2]	0.392	0.486
Maximum	---	---	0.689	0.790

NOTES:

[1] Locations correspond to Fishing Closure Areas (see Figure 1-2).
[2] Only one value available.

SOURCE: New Bedford Harbor Data Base

Table 2-4 summarizes the ranges of whole-body metals concentrations detected in organisms in the New Bedford Harbor area. The tissue residue levels of metals did not show general trends in contaminant concentrations between areas or between species. Overall, cadmium was detected at concentrations lower than either copper or lead. Copper concentrations were highest in crustaceans (i.e., crabs and lobsters), which probably reflects their copper-based heme system.

TABLE 2-4

RANGE [1] OF TOTAL WHOLE-BODY METALS IN NEW BEDFORD HARBOR BIOTA

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

ORGANISM	CADMIUM (ppm)	n [3]	COPPER (ppm)	n [3]	LEAD (ppm)	n [3]
Lobster	0.002NC	2	0.11-24.9	2	0.223-1.29	2
	0.002-0.703	16	20.778-46.814	16	0.106-3.034	16
	0.001-0.538	14	17.997-50.945	14	0.021-1.124	14
	0.002-0.588	21	15.788-62.663	21	0.029-0.842	21
Winter Flounder	0.004-0.014	23	0.692-11.147	23	0.215-3.336	22
	0.002-0.019	27	0.618-19.847	27	0.154-4.523	27
	0.002-0.012	17	0.691-51.642	17	0.099-2.728	17
	0.003-0.099	22	0.480-43.9	22	0.089-6.84	22
Mussel	0.242-0.326	9	1.948-2.49	9	0.293-1.41	9
	0.229-0.271	9	1.895-2.779	9	0.237-1.17	9
	0.326-0.397	6	0.726-0.841	6	0.367-0.647	6
	0.145-0.209	6	0.727-1.081	6	0.134-0.308	6
Quahog	0.087-0.356	18	3.727-8.302	18	0.58-1.901	18
	0.209-0.329	18	1.47-4.055	18	0.488-0.981	18
	0.12-0.381	18	1.302-2.713	18	0.208-3.463	18
	0.119-0.495	10	1.225-2.239	10	0.098-1.720	10
Green Crab	0.075-0.105	5	53.418-262.475	5	4.292-29.768	5
	0.027-0.095	4	12.1-52.897	4	1.45-6.908	4
	0.081[2]	1	201[2]	1	30.6[2]	1
	0.057	3	180.231[2]	3	13.824	3
Polychaetes	NA		NA		NA	
	NA		NA		NA	
	0.065-0.188	6	2.36-6.37	6	0.467-3.979	6
	0.111[2]	3	7.708[2]	3	1.076[2]	3

NOTES:

1 Each value represents the mean of several organisms within one size class

2 Only one value available

3 Total number of organisms sampled in each area

4 Areas correspond to Fisheries Closure Areas

= Not Available

3.0 ECOTOXICITY ASSESSMENT

The ecotoxicity assessment is a two-step process consisting of a compilation and evaluation of available toxicological information, and a synthesis of the information to provide a quantitative assessment of concentration/response data. Available toxicological information, some of which is presented herein, strongly supports the conclusion that PCBs in the marine environment represent a potential threat to biota, and provides additional information necessary to determine the nature and severity of actual or potential adverse effects associated with exposure. Although additional toxicological studies would be useful, the data available are sufficient to allow a quantitative estimation of the risk from contaminant exposure for four of the five groups discussed in Section 2.0. For the remaining group, the polychaete worms, the lack of available data precludes development of good quantitative concentration/response relationships. The concentration/response relationships developed herein will be combined with the exposure concentrations from Section 2.0 to provide the quantitative estimate of risk.

3.1 ECOTOXICITY PROFILES

3.1.1 PCBs

PCBs belong to a class of chemically stable, multi-use industrial chemicals that have been widely distributed in the New Bedford Harbor ecosystem. Electrical component manufacturers in New Bedford used PCBs in transformers and capacitors as dielectric insulating fluids resistant to fire. Discharge of PCBs into the harbor has resulted in contamination of the sediment, water, and biota in the area. Aspects of the structure, fate, and transport of PCBs with importance for determination of ecological risk are discussed in Subsection 1.3.

Adsorption to organic material in sediment is probably the major fate in the marine and estuarine environments of at least the more heavily chlorinated PCBs. Once bound, PCBs may persist for years, with slow desorption providing continuous exposure to the surrounding environment. Because PCBs are persistent in the environment and are lipophilic compounds, they are bioaccumulated (EPA, 1980b). The potential for bioaccumulation of an Aroclor mixture, as with other aspects of the biochemical behavior of PCBs, is related to the percentage of chlorine, with the BCF value generally increasing with higher chlorine content (Callahan et al., 1979). PCBs may be degraded by microorganisms (mainly the mono-, di-, and tri-chlorinated congeners) and by photolysis by ultraviolet light (mainly PCBs with five or more chlorines). Biodegradation rates and mechanisms appear to be specific to individual isomers and it is impossible to generalize about the overall rate for complex mixtures, except that many Aroclors persist for years or decades in the environment. Photolysis is extremely slow, but it may be a significant degradation pathway (EPA, 1980b).

EPA derived an AWQC for the protection of marine organisms for PCBs of 0.03 ug/L (parts per billion [ppb]). This value is based on laboratory-derived BCFs and was established to ensure that PCB burden in edible fish tissue (i.e., the final residue value [FRV]) would not exceed the former FDA tolerance level of 5.0 milligrams per kilogram (mg/kg) and not necessarily to protect ecological receptor organisms (EPA, 1980c). A recalculation of the criteria based on the new tolerance level value of 2.0 mg/kg would establish the new criterion at 0.012 ug/L (ppb); however, this change has not yet been made.

FDA tolerance levels are set to be protective of public health, but are based in part on economical and technical considerations. However, data from acute and chronic toxicity tests using Aroclors indicate that neither acute nor chronic toxicity should occur at the AWQC of 0.03 ug/L.

Marine AWQC, based on final toxicity values, are established to be protective of 95 percent of saltwater species. For PCBs, the AWQC document does not derive final acute or chronic values because determination of acute toxicity concentrations is problematic for PCBs (acute values are often in excess of maximum solubilities); minimum data criteria are not satisfied; and differing toxicities are demonstrated by the various PCB Aroclors and congeners (EPA, 1980b). Therefore, the saltwater AWQC for PCBs is based on the FRV, and is intended to protect the use of marine species as seafood rather than the species themselves, although it is considered sufficiently protective of the organisms as well. As such, these criteria serve as a tool to make general comparisons between the observed water column concentrations in New Bedford Harbor and toxicity information. However, site-specific ecotoxicity data provide a more definitive measure of the potential adverse effects of PCBs to marine organisms in New Bedford Harbor.

Tables B-1, B-2, and B-3 in Appendix B summarize available PCB ecotoxicity data, including acute and chronic toxicity data, as well as bioconcentration data for saltwater species discussed in the toxicological evaluation. Although PCBs have been shown to be acutely toxic to aquatic organisms, the actual exposure concentrations are unknown because the reported concentrations for the acute toxicity tests exceeded solubilities for some portion of PCB isomers, and the complex physical behavior of PCB mixtures makes cross-study comparisons difficult.

Based on the summarized acute and chronic toxicity data on PCBs, marine fish as a group are sensitive to the effects of PCB exposure. Chronic effects observed for marine fish include reduced hatching of embryos, reduced survivorship of fry, lethargy, fin rot, and decreased feeding, as well as mortality. Crustaceans are also quite sensitive, with acute effects being observed at exposures as low as 1 ug/L. The observed effects after chronic exposure for crustaceans include molt inhibition, dispersion of melanin in shells, altered metabolic state, and avoidance (Table B-2). Mortality has also been observed for crustaceans after chronic exposure.

Mollusks as a group are generally not as sensitive to PCB exposure as marine fish and crustaceans; however, reduced growth was observed at an exposure of 5 ug/L. Reduced growth rates are also observed in alga exposed to PCBs. Reduced cell division, reduced carbon dioxide uptake, and even no growth have been observed in alga after chronic exposure to PCBs. When populations of more than one algae species are exposed to PCBs, changes in species ratios and decreased diversity in the communities are observed. Overall PCB toxic effects are varied and at low concentrations. Toxic effects have been reported at concentrations of PCBs higher than the solubilities of the compounds.

BCFs for marine organisms are relatively high, ranging from 800 to greater than 670,000 (EPA, 1980b). Field and Dexter summarized available data for bioaccumulation from PCB-contaminated sediment with ratios ranging to 20 (Field and Dexter, 1988). These high factors would be predictable based on the lipophilic nature of PCBs. BCFs vary depending on several factors, including the level of total organic carbon (TOC) in the sediment and the length of exposure. BCFs vary among species and for different congeners. In general, the factors will be higher for species with greater amounts of fatty tissue. For congeners, the highest factors appear to occur among the congeners with five and six chlorine atoms; the lowest among those with eight and nine atoms (Lake et al., 1989).

3.1.2 Copper

Copper is a necessary nutrient for plants and animals; however, it is toxic at higher concentrations (EPA, 1985a). The copper ion is highly reactive and complexes with many inorganic and organic constituents of natural waters (EPA, 1985a). Hydrous iron and manganese oxides can effectively remove almost all free copper from the water column (Lee, 1975); and sediment/clay complexes, carbonates, and organic acids are all similarly effective under particular conditions. Most organic and inorganic copper complexes and precipitates appear to be much less toxic than free cupric ion.

Relatively few marine toxicological data are available for copper. However, mollusks and phytoplankton appear to be most sensitive to copper. Tables B-4 and B-5 in Appendix B summarize the toxicity data available for marine organisms. Copper has been shown to be acutely toxic to embryos of the blue mussel (Mytilus edulis) at 5.8 ug/L (Martin et al., 1977), and several diatom and marine alga species are sensitive to copper in the 1-to-10-ppb range. In fact, copper has been historically used as an aquatic herbicide and as a molluscicide to control schistosomiasis. Mean lethal concentration (LC_{50}) values for tests on winter flounder embryos (Pseudopleuronectes americanus) and the American lobster (Homarus americanus) were 130 and 69 ug/L, respectively (EPA, 1985a).

The only chronic data available for marine organisms are for Mysidopsis bahia; EPA established a chronic value of 54 ug/L based on lifecycle tests with this species. Various phytoplankton, polychaete worms, and mollusks have been shown to bioaccumulate copper with BCF values ranging from less than 100 to over 20,000. The marine chronic AWQC was established by EPA at 2.9 ug/L (ppb).

3.1.3 Cadmium

Although cadmium is insoluble in water, its chloride and sulphate salts readily solubilize. Humic acids and, to a lesser extent, hydrous iron and manganese oxides, appear to be primarily responsible for determining the extent of adsorption to sediment, while increased acidity and oxygenation tends to amplify desorption rates and subsequent bioavailability (Eisler, 1985; and Forstner, 1983). In addition, increasing salinity appears to mitigate the toxicological impact of this contaminant (EPA, 1985b). Tables B-6 and B-7 in Appendix B summarize the available saltwater ecotoxicity data for cadmium.

In general, freshwater species are considerably more sensitive to cadmium poisoning than marine species (Eisler, 1985). Among marine organisms, invertebrates are most sensitive to cadmium toxicity, with acute test results ranging from 41 to 135,000 ug/L for Mysidopsis bahia and an oligochaete worm, Monophylephorus cuticalcatus, respectively (EPA, 1985b).

Sublethal effects, including growth retardation, physiological disruptions, and alteration of oxygen consumption and respiratory rates, have been observed in marine organisms exposed to ambient cadmium concentrations on the order of 0.5 to 10 ug/L (Eisler, 1985).

Marine organisms can readily bioconcentrate cadmium, and BCF values over 2,000 have been recorded in some polychaete worms and mollusks (EPA, 1985b). However, reported BCFs for the lobster (Homarus americanus) and a marine fish, Fundulus heteroclitus, were 21 and 15, respectively (Eisler, 1985). EPA derived a chronic AWQC of 9.3 ug/L for the protection of marine organisms for cadmium.

3.1.4 Lead

Lead is most soluble under aqueous conditions characterized by low pH, low organic content, low particulate matter, and low concentrations of the salts of calcium, cadmium, iron, manganese, and zinc (Eisler, 1988). Most lead entering aquatic environments is quickly precipitated to bed sediments, and is released only under specific conditions (Demayo et al., 1982).

Relatively few toxicological data for marine species are available, with chronic-level effects observed in some organisms, particulary phytoplankton, in the 1-to-10-ug/L range. The plaice, Pleoronectes platessa, was acutely sensitive to tetramethyl lead at 50 ug/L (Eisler, 1988); a lifelong maximum acceptable toxicant concentration (MATC) between 17 and 37 ug/L was calculated for Mysidopsis bahia.

BCFs for lead in marine organisms ranged from 17.5 to 2,570 for the quahog (Mercenaria mercenaria) and the blue mussel (Mytilus edulis), respectively (EPA, 1980b). However, there is no evidence to indicate that lead is transferred through aquatic food chains (Eisler, 1988).

Tables B-8 and B-9 in Appendix B summarize available ecotoxicological data specific to the effects of lead exposure to marine organisms. Based on these data, EPA derived a chronic AWQC of 5.6 ug/L for the protection of marine organisms for lead.

3.2 EFFECTS EVALUATION

3.2.1 Methods

PCB and metals effects curves were constructed for the four taxonomic groups (i.e., marine fish, crustaceans, mollusks, and alga) for which ecotoxicity data were available. Data on benchmark effects were summarized, and the mean and variance of these data were used in the joint probability analysis to estimate risk, and to generate cumulative frequency probability curves. The curves provide an evaluation of probability of effect at various contaminant concentrations.

The standard acute benchmark for evaluating the acute response of an aquatic organism to the environmental concentration of a toxic contaminant is the 96-hour median LC_{50} (EPA, 1982; and ASTM, 1984). However, for purposes of risk assessment, the acute benchmark is not appropriate because the organisms are assumed to be exposed for periods longer than 96 hours. A more appropriate benchmark is the MATC, which is the threshold for significant effects on growth, reproduction, or survival (EPA, 1982; and ASTM, 1984). The benchmark is based on the most sensitive response of the organism to the contaminant in question.

Few MATC data are available for marine organisms, and the research that has been performed is limited with respect to both contaminant type and test organisms used. There are insufficient MATC data for PCBs to generate distributions for any of the taxonomic groups of interest. For this risk assessment, MATCs for the four taxonomic groups were developed using a method described by Suter and Rosen (Suter et al., 1986; and Suter and Rosen, 1986). This method uses an errors-in-variables regression model to predict a toxicological endpoint (in this case, the MATC) based on an extrapolation from existing endpoints for similar organisms. The regression equations used were established based on several large aquatic toxicological data bases (Suter and Rosen, 1986). For example, the model allows extrapolation from the LC_{50} of one species to the LC_{50} of another; similar extrapolations can be performed between LC_{50}s and MATCs. Therefore, a regression equation can be developed that has a coefficient (slope) and constant (intercept) that characterizes a between-taxon LC_{50} relationship or a within-taxon relationship between LC_{50}s and MATCs.

The errors-in-variables approach considers the following characteristics of toxicity data that a linear least-squared model would not address: (1) the observed values of both the independent (X) and dependent (Y) variables have inherent variability and are subject to measurement error; (2) the independent variable is not a controlled variable; and (3) the values assumed by (X) and (Y) are open-ended and non-normally distributed (Ricker, 1973). This method allows for quantification of uncertainty from interspecific differences in sensitivity, and the variability of the relationship between acute and chronic effects of contaminants. The uncertainty is quantified in the variances that result from the extrapolation. This variance is then applied in the joint probability analysis, which uses the estimated toxicological benchmark value and its variance, along with an EEC and its variance to estimate risk of chronic effects to a particular group of organisms. The final risk estimate is interpreted as the probability of an adverse effect being realized in a typical member of the group in question, given the variability in contaminant levels.

This model and its application are discussed in more detail in Section 4.0. MATCs for four groups of organisms (i.e., marine fish, crustaceans, mollusks, and alga) representative of the range of organisms found in New Bedford Harbor were developed using this approach. The taxonomic groupings were necessary to facilitate the application of the errors-in-variables methodology, because extrapolations are within or between taxonomic levels. A comparable analysis by strict trophic and/or habitat classification by this method would not have been possible because multiple taxa groups would be a part of such an analysis. However, these groups generally also define a primary means of exposure (e.g., via water or sediment) and, therefore, allow consistency with respect to applying exposure concentrations to provide a risk estimate.

For marine fish, crustaceans, and mollusks, MATCs were developed using the errors-in-variables methodology. For the algae, a chronic effect concentration was developed based on the existing toxicological data. The data used for the overall MATC development for alga and mollusks came from the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler, 1986). These data sets were also used as the source of the LC_{50} for the sheepshead minnow and the MATC for Daphnia magna used in extrapolations for marine fish and crustacean MATCs.

All data used for the regressions were log-transformed. Test results reported as greater than or less than a particular value were not used. When replicate data were available for a chemical-species pair, the geometric mean for the species was used. Use of the geometric rather than the arithmetic mean for replicate tests is consistent with EPA methods for AWQC development (EPA, 1982).

3.2.2 Application and Results

3.2.2.1 Marine Fish

Development of the MATCs for marine fish was based on previously reported relationships. Suter and Rosen performed extrapolations between the LC_{50}s for sheepshead minnow (*Cyprinodon variegatus*) and LC_{50}s for marine species, as well as derivation of the errors-in-variables relationship between marine fish LC_{50} and marine fish MATCs (Suter and Rosen, 1986). The slope, intercept, and variance from these extrapolations used in the MATC development and risk assessment for marine fish in New Bedford Harbor are presented in Table 3-1.

The overall marine fish MATC for PCBs was created by a double extrapolation: first from the sheepshead minnow chronic LC_{50} for PCBs (0.93 ug/L) to a typical marine fish LC_{50} for PCBs

TABLE 3-1
PCB MATC ESTIMATES FOR ORGANISMS AT NEW BEDFORD HARBOR

NEW BEDFORD HARBOR
ECOLOGICAL RISK ASSESSMENT

TAXON	SLOPE	INTERCEPT	MATC	TOTAL VARIANCE
Marine Fish	0.97	0.03		
	0.98	-0.6	-0.601	1.021
Crustaceans	0.95	0.0	0.668	0.956
Mollusks	1.577	-0.456		
	0.98	-0.6	1.358	3.024
Algae			0.987	4.907

NOTES:

1. The basic regression equation that defines the extrapolation is Y = Intercept + (X * Slope), where X is the acute toxicological estimate and Y the extrapolated MATC value.
2. No extrapolation was done for algae; rather, chronic data were used to estimate the benchmark value for the taxon.
3. In cases where two sets of slope and intercept values are listed, the first set is for a LC50-to-LC50 extrapolation, and the second for the final LC50-to-MATC extrapolation.
4. All units expressed as Log (base 10) ug/L.

(0.99 ug/L), then to a marine fish MATC of 0.25 ug/L. The chronic LC_{50} value used as the starting point for these extrapolations was an early life stage test using Aroclor 1254. Similar testing with Aroclor 1016 produced similar responses only at concentrations above 10 ug/L. Other Aroclors are expected to fall generally within this range, and the lower value for Aroclor 1254 provides a conservative estimate of the toxicity of the actual mix of PCB congeners in New Bedford Harbor. The effect curve, which is a cumulative probability plot based on the MATC value and its variance, is shown in Figure 3-1.

Approximately 95 percent of the calculated MATC values for marine fish falls within a range of four orders of magnitude; chronic values in the literature, most of which are based on one of three species, span approximately half this range. This difference is largely a result of the procedure that uses the actual data as a sample from the universe of MATCs and generates a probability plot for all marine species in the taxon of interest. The actual range for species residing in New Bedford Harbor may well be smaller; however, there is no way of developing such a site- specific MATC with the available data.

The metal MATC values for marine fish were extrapolated using a relationship between the MATCs of the mysid, Mysidopsis bahia and the MATCs of fish developed by Suter and Rosen (Suter and Rosen, 1986). The extrapolations were from the mysid MATCs of 54, 5.5, and 25 ug/L for copper, cadmium, and lead, respectively. The MATCs derived for marine fish were 329, 32, and 150 ug/L for copper, cadmium, and lead, respectively.

The MATC effects curves are shown in Figures B-1, B-2, and B-3 in Appendix B. The slope, intercept, and variance from these extrapolations used in the MATC development and risk assessment for metals and marine fish in New Bedford Harbor are presented in Tables B-10, B-11, and B-12.

3.2.2.2 Crustaceans

The PCB MATC for crustaceans was obtained from the association between the MATC for the cladoceran (Daphnia magna) and MATCs for marine crustaceans developed by Suter and Rosen (Suter and Rosen, 1986). The slope, intercept, and variance developed in this errors-in-variables model are presented in Table 3-1. One extrapolation from the cladoceran MATC (5.14 ug/L) was required to derive the typical marine crustacean MATC of 4.66 ug/L. The MATC probability curve for crustaceans is shown in Figure 3-1.

A single extrapolation was required to develop the metal MATCs for crustaceans. These MATC values were extrapolated using a relationship between the MATCs of the mysid, Mysidopsis bahia, and the MATCs of crustaceans developed by Suter and Rosen (Suter

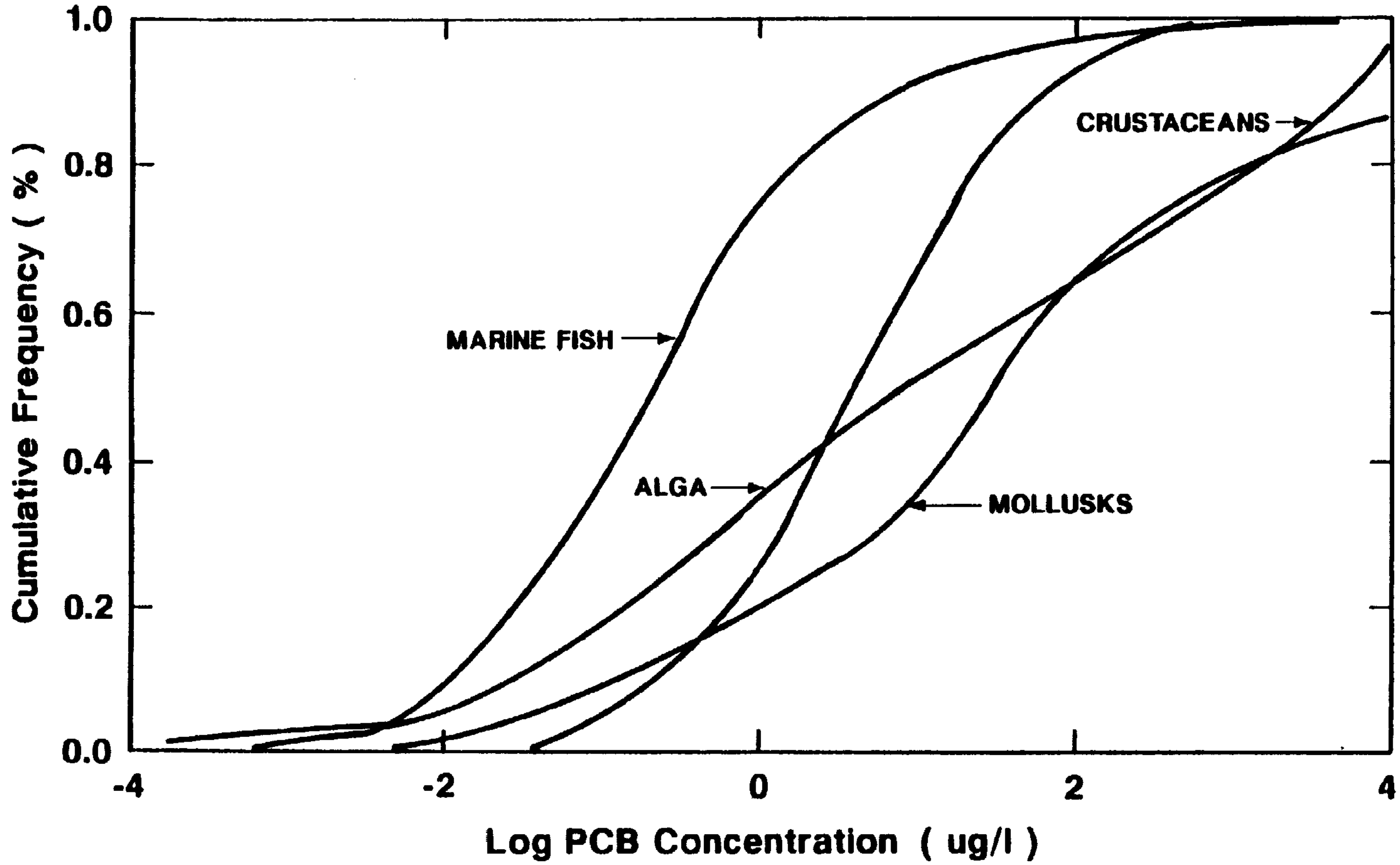


FIGURE 3-1
MATC CURVES FOR PCBs
NEW BEDFORD, MASSACHUSETTS

and Rosen, 1986). The extrapolations were from the mysid MATC values of 54, 5.5, and 25 ug/L for copper, cadmium, and lead, respectively. The extrapolated MATCs developed for crustaceans were 65.5, 10.5, and 35.3 ug/L for copper, cadmium, and lead, respectively. The slope, intercept, and variance from these models are shown in Tables B-10, B-11, and B-12 in Appendix B. The MATC curves for copper, cadmium, and lead are shown in Figures B-1, B-2, and B-3, respectively.

3.2.2.3 Mollusks

To develop the PCB MATC for mollusks, two extrapolations were needed. First, a relationship between the LC_{50}s for the mysid, Mysidopsis bahia, and LC_{50}s of mollusks was developed. The relationship between these species was used because the greatest number of matches between chemical-species pairs was available and, although there is no close taxonomic relationship, the mysid is a standard test species. Because there are no MATC data available for mollusks, an estimate of the MATC was performed by using the relationship between marine fish LC_{50}s and MATCs, on the assumption that the ratios between acute and chronic effects for marine fish and mollusks are similar. The slopes, intercepts, and variances used in this MATC development are shown in Table 3-1.

The mollusk LC_{50} of 99.61 ug/L was obtained by forward extrapolation from the mysid LC_{50} (36.0 ug/L). The estimated mollusk LC_{50} was then used to estimate the typical mollusk MATC (22.82 ug/L) based on the LC_{50}/MATC relationship for marine fish. The effects curve is shown in Figure 3-1. There is a large variance associated with this MATC due to the double extrapolation. Large variances were observed by Suter and Rosen for similar extrapolations between higher level taxonomic groups (Suter et al., 1986; and Suter and Rosen, 1986). Because the variance for the extrapolation from LC_{50} to MATC for marine fish is small, its use in this application may result in an underestimation of the variance associated with the MATC for mollusks.

As in the case of PCBs, limited data are available on metal MATCs for mollusks. To develop MATCs for mollusks, the same marine fish LC_{50}-to-MATC relationship was used as for PCBs, assuming that the ratios between acute and chronic effects for marine fish and mollusks are similar. The LC_{50}s used in this extrapolation were developed from values reported in the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler 1985 and 1986). These data are compiled in Tables B-4 through B-9 in Appendix B. For each metal, the mollusk LC_{50} value used in the extrapolation is a geometric mean of the values reported for all mollusks.

The metal MATCs for mollusks were derived from the mollusk LC_{50} values of 72.4, 2,666, and 1,244 ug/L for copper, cadmium, and lead, respectively. The single forward extrapolation for each metal estimated the mollusk MATCs to be 16.7, 571, and 271 ug/L for copper, cadmium, and lead, respectively. The effects curves for the MATCs are presented in Figures B-1, B-2, and B-3 in Appendix B. The slope, intercept, and variance from these extrapolations are presented in Tables B-10, B-11, and B-12.

3.2.2.4 Polychaetes

There were sufficient acute toxicological data for the three metals to develop MATC estimates for polychaetes, using the crustacean LC_{50} and MATC extrapolation developed by Suter and Rosen (Suter and Rosen, 1986). In this case, it was assumed that the ratios between acute and chronic effects for crustaceans and polychaetes are similar. The LC_{50}s used in this extrapolation were developed from values reported in the AWQC and Eisler documents (EPA, 1980a, 1980b, and 1980c; and Eisler 1985 and 1986). Tables B-4 through B-9 in Appendix B summarize of the toxicological data used to develop MATC estimates for polychaetes. The polychaete LC_{50} for each metal is a geometric mean of the values reported for all polychaetes and oligochaetes.

The metal MATCs for polychaetes were derived from the polychaete LC_{50} values of 199, 9,682, and 10,691 ug/L for copper, cadmium, and lead, respectively. A single forward extrapolation for each metal was necessary to estimate the polychaete MATCs as 30.2, 1,276, and 1,409 ug/L for copper, cadmium, and lead, respectively. MATC curves for copper, cadmium, and lead are shown in Figures B-1, B-2, and B-3, respectively. The slope, intercept, and variance from these individual extrapolations are presented in Tables B-10, B-11, and B-12.

3.2.2.5 Algae

For the algal species at the New Bedford Harbor site, a benchmark concentration was developed using the geometric mean of the results from chronic tests as presented in the AWQC and Eisler documents (EPA, 1980; and Eisler, 1986). Although this value is not an MATC by definition, it is a reasonable best estimate of chronic toxicological effects of PCBs on algal species based on the limited data available. The benchmark concentration of 9.71 ug/L has a high amount of variance (4.44); this is due to the large amount of variability in reported responses to PCBs. The effects curve is shown in Figure 3-1.

For the metals, a geometric mean was developed from chronic effects data presented in the AWQC and Eisler documents (EPA, 1980a and 1980c; and Eisler, 1985 and 1988). The benchmark values derived were 12, 99.3, and 234 ug/L for copper, cadmium, and lead, respectively. The effects curves for the MATCs are shown in Figures B-1, B-2, and B-3 in Appendix B. Summary statistics for these benchmark concentrations are in Tables B-10, B-11, and B-12.

3.2.3 Evaluation of MATCs

Because of the limited amount of data available about the effects of PCBs and metals on marine organisms, the estimates of MATC or chronic effect benchmarks as used in this risk assessment have some uncertainty, which was quantified to some extent by the variances from the errors-in-variables extrapolations. The relative effect of this source of uncertainty may be observed graphically by comparison of the slope of the probability function for the MATC of each group in Figure 3-1. This uncertainty is also evident in the effect of the variance on results of the analysis of extrapolation error model used for risk characterization in Section 4.0. In all cases, the variance in the estimates for metal MATC values was not as high as for PCBs, primarily due to the fact that only one extrapolation was necessary.

Another area of uncertainty for these MATC estimates results from the need to perform extrapolations from a single species to a taxonomic group consisting of many species, some of which may be only distantly related. If the single species used in the extrapolation happens to be particularly sensitive to contaminants, the final estimate of the group MATC may be overly conservative. This is probably the case for the extrapolation from the sheepshead minnow to marine fish in general. The PCB LC_{50} for the sheepshead minnow (0.93 ug/L), the species used to develop most of the available data, is quite low, driving the marine fish MATC to a lower value than may be the case. However, other marine fish tested also have low LC_{50}s for PCBs.